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Human and hospital-use pharmaceuticals, aquaculture medicines and personal care products

Environmental Screening of Selected Organic Compounds 2008

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Norwegian Institute for Water Research

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Human and hospital-use pharmaceuticals, aquaculture medicines and personal care products

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Foreword

On behalf of the Norwegian Pollution Control Authority (SFT) the Norwegian Institute for Air Research (NILU), Norwegian Institute for Water Research (NIVA), and Swedish Environmental Research Institute (IVL) have analyzed selected organic compounds which are used in human and aquaculture pharmaceuticals and in personal care products. These include samples from municipal and hospital wastewater/sludge, surface water, sediment, and blue mussel taken in 2008 from selected wastewater treatment plants (WWTP) and marine sites. The results of this study are presented in this report.

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Silje Klausen: Design of the result figures.

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Christian Vogelsang: Sampling and handling of samples from wastewater treatment plants, and responsible for assessment of results from wastewater treatment plants and freshwater.
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Norwegian Pollution Control Authority (SFT)
Bård Nordbø: Project coordinator at SFT.

NILU, Oslo, 24 April 2009

Martin Schlabach
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Content

1. Summary

Background
On behalf of the Norwegian Pollution Control Authority (SFT), the Norwegian Institute for Air Research (NILU), the Norwegian Institute for Water Research (NIVA), and the Swedish Environmental Research Institute (IVL) monitored pharmaceuticals, hospital-use pharmaceuticals, aquaculture medicines and personal care products in samples from hospital effluent water, wastewater treatment facilities, seawater, marine sediment, and blue mussels in samples collected in 2008 as a part of a screening.

The survey covers eleven pharmaceuticals, seven hospital antibiotics, three x-ray contrast agents, five cytostatic agents (and two metabolites), eight personal care products, and seven aquaculture medicines in various environmental samples. The aquaculture medicines were analysed in samples collected from two fish farms in Western Norway. The remaining analytes were analysed in samples collected from greater Oslo and Tromsø. The Oslo samples were effluent water from Ullevål hospital and VEAS, receiving water, sediment and biota from the inner Oslofjord. The Tromsø samples included effluent water from University Hospital in Northern Norway (UNN) and effluent samples from Breivika sewage treatment plant (STP), receiving water, sediment and biota in Tromsøsund.

Results
Pharmaceuticals
Analysis included eleven pharmaceutical compounds: amitriptyline, atorvastatin, carbamazepine, morphine, naproxen, paracetamol, propranolol, sertraline, spiramycin, tamoxifen, and warfarin.

Tamoxifen was the only compound from this group found in biota. Amitriptyline, carbamazepine, morphine, naproxen, and propranolol were all detected in surface water. All analytes, apart from tamoxifen, were detected in the STP effluents. Amitriptyline, atorvastatin, carbamazepine, naproxen, propranolol, sertraline, tamoxifen, and warfarin were detected in sludge. Atorvastatin, paracetamol, sertraline, and warfarin were not detected in receiving waters, sediments or mussels.

Hospital-use pharmaceuticals
A hospital-use pharmaceutical is exclusively used in hospitals. The antibiotics amoxicillin, cefotaxime, cefalotin, meropenem, ofloxacin, penicillin G, pivmecillinam, the x-ray contrasting agents iohexol, iodixanol, iopromide, and the cytostatics doxorubicin, irinotecan, bortezomib, docetaxel, paclitaxel, (and the metabolites doxorubicinol and 6-OH-paclitaxel) were included for analysis.

Cefotaxime was detected in hospital effluents, and in STP effluent water. Ofloxacin was detected once in an effluent sample. Amoxicillin, cefotaxime, cefalotin, meropenem, ofloxacin, penicillin G, and pivmecillinam were not detected in any receiving water, sediment or mussel samples in this screening.

Iodixanol, iopromide, and iohexol were all detected in surface water. Iodixanol, iohexol, and iopromide were detected in sediment. These compounds were not analysed in biota samples. All compounds were detected in hospital effluents and in STP effluent water, with the concentrations in Tromsø being more than 10 times higher. Little or no loss of the analytes

was observed upon STP passage. Iohexol and iopromide were not detected in sludge whereas iodixanol was detected in sludge.

Irinotecan was detected in hospital effluent water and in STP effluent water. The metabolite 6-OH-paclitaxel was detected in STP effluent water. Irinotecan and 6-OH-paclitaxel were not detected in any receiving water, sediment or mussel samples. No other cytostatics (bortezomib, docetaxel, doxorubicin and doxorubicinol, and paclitaxel) were detected in any sample.

Aquaculture medicines

The aquaculture medicines cypermethrin, deltamethrin, emamectin, fenbendazole, flumequine, oxolinic acid, and praziquantel, were analysed in surface water, sediment, and blue mussel in close proximity to two fish farms.

No analytes were detected in blue mussel. Emamectin was detected in the sediment at both fish farms. Oxolinic acid was detected in surface water at both fish farms. Oxolinic acid was also detected in the sediment, at lower concentration at fish farm 1 than at fish farm 2. The other studied aquaculture medicines were not detected.

Personal care products

Avobenzone, butyl paraben, cetrimonium, cocoamidopropyl betaine, diethylphthalate (DEP), EDTA, sodium dodecyl sulfate (SDS), and sodium laureth sulfate (SDSEO) are high volume personal care products.

Avobenzone was not detected in any sample. Butyl paraben was detected in effluent and receiving water. Butyl paraben was not detected in any sediment, biota or sludge sample. Cetrimonium was detected in effluent water, sludge, sediments, and blue mussels. Cocoamidopropyl betaine was only detected in sludge samples. Biota samples were not analysed for cocoamidopropyl betaine. DEP was detected in effluent water, sludge, receiving water, in blue mussels, and in sediment. EDTA was detected in effluent water, sludge, receiving water and in sediment. SDS was detected in effluent water, sludge, receiving water; biota samples were not analysed for SDS. Sodium laureth sulfate (SDSEO) was detected in effluent waters, sludge, and receiving waters. Biota samples were not analysed for SDSEO.

Risk assessment of the results

The relevance of the results, i.e. if they cause environmental concerns is evaluated by the following set of criteria:

(i) If the compound was not detected or only detected in waste water, the compound was assessed to be of no or little environmental concern.

(ii) For compounds detected in receiving water and/or sediment, its highest detected concentration was compared with the worst case ecotoxicological effect concentration found in the scientific literature:

   a. If the difference between highest observed concentration and the worst case ecotoxicological effect concentration found in the scientific literature was more than 100 000, the compound was assessed to be of little or no environmental concern.
   b. If the difference between highest observed concentration and the worst case ecotoxicological effect concentration found in the scientific literature was

more than 1 000, but less than 100 000, the compound was assessed to be of some environmental concern.

c. If the difference between highest observed concentration and the worst case ecotoxicological effect concentration found in the scientific literature was less than 1000, the compound was assessed to be of environmental concern. 1000 was chosen as a safety factor as this often is applied as a safety factor in environmental risk assessments

(iii) Compounds identified in biota are automatically of environmental concern.

Conclusions

Based on this simple risk assessment, the compounds are classified as following:

No environmental concern:

General pharmaceuticals: amitriptyline, atorvastatin, paracetamol, sertraline, spiramycin, and warfarin;

Hospital-use pharmaceuticals: amoxicillin, cefotaxime, cefalotin, meropenem, ofloxacin, penicillin G, pivmecillinam, the x-ray contrasting agents iohexol, iodixanol, iopromide, and the cytostatics doxorubicin, irinotecan, bortezomib, docetaxel, paclitaxel, (and the metabolites doxorubicinol and 6-OH-paclitaxel);

Aquaculture medicines: cypermethrin, deltamethrin, emamectin, fenbendazole, flumequine, oxolinic acid, and praziquantel;

Personal care products: avobenzone and cocoamidopropyl betaine.

Some environmental concern:

General pharmaceuticals: Tamoxifen and morphine;

Personal care products: EDTA, butyl paraben, sodium dodecyl sulphate (SDS), and sodium laureth sulphate (SDSEO).

Environmental concern:

General pharmaceuticals: carbamazepine, naproxen, propranolol;

Personal care products: cetrimonium, and diethyl phthalate.

For compounds which are categorized as of some environmental concern or of environmental concern, toxic and other adverse effects on aquatic organisms and on the aquatic environment cannot be excluded. The environmental levels and effects of these compounds should therefore be studied in more detail.

2. Sammendrag

På vegne av Statens forurensningstilsyn (SFT) har Norsk institutt for luftforskning (NILU), Norsk institutt for vannforskning (NIVA) og Svenska miljöinstitutet (IVL) monitorert legemidler, sykehusfarmasøytika, veterinærmedisiner og personlige pleieprodukter i prøver fra avløpsvann fra sykehus og kloakkrenseanlegg, slam, sjøvann, marine sedimenter og blåskjell. Prøvene ble hentet i 2008 i et screeningprosjekt finansiert av SFT.

Undersøkelsen dekker elleve legemidler, syv sykehusspesifikke antibiotika, tre røntgenkontrastmidler, fem cytostatika (og to metabolitter av disse), syv personlig pleieprodukter og syv veterinærmedisiner tatt i ulike miljøprøver. Veterinærmedisinene har blitt analysert i prøver tatt ved to oppdrettsanlegg på Vestlandet og Nordvestlandet. De øvrige analyttene har blitt analysert i prøver som er tatt i stor-Oslo eller Tromsø. Prøvene fra stor-Oslo var fra avløpsvann fra Ullevål universitetssykehus og behandlet (utløp) vann fra VEAS kloakkrenseanlegg, videre ble prøver av resipientvann, sediment og blåskjell tatt fra indre Oslofjord. Prøvene fra Tromsø var fra avløpsvann fra Universitetssykehuset i Nord-Norge (UNN) og behandlet avløpsvann fra Breivika kloakkrenseanlegg, videre ble prøver av resipientvann, sediment og blåskjell tatt i Tromsøsund.

Resultater
Legemidler
Analysene omfattet de elleve forbindelsene amitriptylin, atorvastatin, karbamazepin, morfin, naproksen, paracetamol, propranolol, sertralin, spiramycin, tamoksifen og warfarin.

Tamoksifen var den eneste forbindelsen fra denne gruppen som ble påvist i blåskjell. Amitriptylin, karbamazepin, morfin, naproksen og propranolol ble alle påvist i resipientvann. Alle unntatt tamoksifen ble påvist i avløpsvann fra kloakkrenseanlegg. Amitriptylin, atorvastatin, karbamazepin, naproksen, propranolol, sertralin, tamoksifen og warfarin ble alle detektert i slam..

Sykehusfarmasøytika
Et sykehuslegemiddel benyttes (nesten) utelukkende på sykehus. Analysene omfattet antibiotikaene amoksicillin, cefotaksim, cefalotin, meropenem, ofloksacin, penicillin G (benzylpenicillin), pivmecillinam, røntgenkontrastmidlene iodixanol, joheksol og jopromid, og cytostatikaene bortezomib, docetaxel, doksorubicin, irinotecan og paclitaxel, samt metabolittene doksorubicinol og 6-OH-paclitaxel.

Cefotaksim ble påvist i avløpsvann fra sykehus og i avløpsvann fra kloakkrenseanlegg. Ofloksacin ble påvist i avløpsvann fra sykehus. Amoksicillin, cefotaksim, cefalotin, meropenem, ofloksacin, penicillin og pivmecillinam ble ikke påvist i noen overflatevann, sediment eller blåskjellprøver i denne screeningen.

Iodixanol, jopromid og joheksol ble alle påvist i resipientvann og sediment. Forbindelsene ble ikke analysert i biotaprøver. Alle forbindelsene ble påvist i avløpsvann fra sykehusene og kloakkrenseanlegg. Det ble observert liten eller ingen eliminasjon av disse forbindelsene i kloakkrenseanleggene. Iodixanol ble funnet i slam.

Irinotecan ble påvist i avløpsvann fra sykehus og kloakkrenseanlegg. Metabolitten 6-OH-paclitaxel ble påvist i avløpsvann kloakkrenseanlegg. Irinotecan og 6-OH-paclitaxel ble ikke

påvist i overflatevann, sediment eller blåskjell. Ingen andre cytostatika (bortezomib, docetaxel, doksorubicin, doxorubicinol og paclitaxel) ble påvist i noen prøver.

Akvakulturmedisiner
Akvakulturmedisinene cypermetrin, deltametrin, emamektin, fenbendazol, flumekvin, oksolinsyre og prazikvantel ble analysert i overflatevann, sediment og blåskjell i nærheten av to fiskeoppdrettsanlegg.

Ingen veterinærmedisiner ble påvist i blåskjell. Emamektin ble funnet i sediment ved begge anleggene. Oksolinsyre ble funnet i overflatevann ved begge anleggene, og ble også påvist i sediment i lavere konsentrasjon ved anlegg 1 enn anlegg 2. De andre undersøkte akvakulturmedisiner ble ikke påvist.

Personlig pleieprodukter
Avobenzon, butylparaben, cetrimonium, cocoamidopropylbetain, dietylftalat (DEP), EDTA, natriumdodekylsulfat (SDS), natrium lauretsulfat [lauryl(poly)etersulfat; SDSEO] er personlig pleieprodukter som benyttes i store volum.

Avobenzone ble ikke påvist i noen prøver. Butylparaben ble påvist i resipientvann og i avløpsvann fra kloakkrenseanlegg. Butylparaben ble ikke funnet i noen sediment, blåskjell eller slamprøver. Cocoamidopropylbetain ble kun funnet i slamprøver. Cetrimonium ble påvist i sediment, blåskjell og avløpsvann kloakkrenseanlegg samt i slam. Dietylftalat (DEP) ble påvist i resipientvann, blåskjell, sediment, avløpsvann fra kloakkrenseanlegg, samt i slam. EDTA ble påvist i resipientvann og i sediment, samt i avløpsvann fra kloakkrenseanlegg og slam. SDS ble påvist i overflatevann, avløpsvann fra kloakkrenseanlegg og slam. SDSEO ble påvist i resipientvann, avløpsvann og slam.

Risikovurdering av resultatene
Relevansen av resultatene, dvs. hvorvidt de er gjenstand for miljømessig bekymring ble evaluert etter følgende kriterier:

(i) Dersom forbindelsen ikke ble detektert eller kun detektert i avløpsvann og/eller slam, ble forbindelsen vurdert å være gjenstand for ingen eller liten miljømessig bekymring.
(ii) For forbindelser som ble detektert i overflatevann og/eller sediment, ble den høyeste påviste konsentrasjonen sammenlignet med den verste bestemte økotoksisitetskonsentrasjonen i den vitenskaplige litteraturen:
    a. Dersom forskjellen mellom den høyeste påviste konsentrasjonen og den verste bestemte økotoksisitetskonsentrasjonen var større enn 100 000, ble forbindelsen vurdert å være av liten eller ingen miljømessig bekymring.
    b. Dersom forskjellen mellom den høyeste påviste konsentrasjonen og den verste bestemte økotoksisitetskonsentrasjonen var større enn 1 000, men mindre enn 100 000, ble forbindelsen vurdert til å være av en viss miljømessig bekymring.
    c. Dersom forskjellen mellom den høyeste påviste konsentrasjonen og den verste bestemte økotoksisitetskonsentrasjonen var under 1 000, ble forbindelsen vurdert å være av miljømessig vurdering. 1 000 ble valgt som sikkerhetsfaktor da dette ofte blir anvendt innen miljørisikovurderinger.
(iii) Forbindelser som ble funnet i biota ble automatisk vurdert å være av miljømessig bekymring.

Konklusjon
Basert på denne enkle risikovurderingen, ble forbindelse som er inkludert i denne screeningen klassifisert følgende:

Liten eller ingen miljømessig bekymring:
Legemidler: Amitriptylin, atorvastatin, paracetamol, sertralin, spiramycin og warfarin.
Sykehusfarmasøytika: Amoxicillin, cefotaksim, cefalotin, meropenem, ofloksacin, penicillin G, pivmecillinam, iodixanol, johexol, jopromide, doxorubicin, irinotecan, bortezomib, docetaxel, paclitaxel, (og metabolittene doxorubicinol og 6-OH-paclitaxel).
Akvakulturmedisiner: Cypermetrin, deltametrin, emamektin, fenbendazole, flumequine, oksolinsyre og praziquantel.
Personlig pleieprodukter: Avobenzon og cocoamidopropylbetain.

Noe miljømessig bekymring:
Legemidler: Tamoksifen og morfin.
Personlig pleieprodukter: EDTA, butylparaben, laurylsulfat og lauretsulfat.

Miljømessig bekymring.
Legemidler: Karbamazepin, naproksen og propranolol.
Personlig pleieprodukter: Cetrimonium og dietylftalat.

For disse stoffer kan toksiske og andre effekter på vannlevende organismer og det akvatiske miljøet ikke utelukkes. Både forekomst og effekter av disse stoffer bør undersøkes og kartlegges bedre.

3. Background and purpose

3.1 General

Many tonnes of human pharmaceuticals and aquaculture medicines are sold in Norway every year and the personal care products market is worth several billion NOK a year. Most of these xenobiotic compounds and their metabolites end up in rivers, streams and fjords via the sewage system. The environmental risk these substances pose to the environment is not clear. Acute environmental risk assessments suggest a few examples where the environment is at risk, however due to the specific mechanisms of these biologically active substances the chronic long-term risks are less clear. Environmental monitoring is therefore important to better understand the fate and occurrence of these substances to allow better risk assessment and environmental protection. On the other hand incorporating chronic Ecotoxicological effects testing of aquatic life into assessment strategies is an important step toward increased understanding of environmental effects.

Pharmaceuticals and personal care products (PPCP) are, as the acronym suggests, often treated together, although there are differences. Pharmaceuticals are used almost exclusively to treat an unwanted (pathologic) condition, except for x-ray contrasting agents and other diagnostics, and they are developed to have a highly specific biological (or biocide) effect. Personal care products contain compounds useful for their intended cosmetic rather than their biological effect. In fact, most personal care products are claimed to be biological inert. The environmental concerns regarding personal care products are due to their high-volume use and for several compounds due to their reported ecotoxicological effects.

One common feature of PPCPs is that they are transported with the sewage system. If they are not efficiently removed at an STP, they are discharged into receiving waters. One exception here is aquaculture medicines that are used to treat the fish in situ, and the excess is discharged into the receiving waters.

There is also a difference at governmental level. Pharmaceuticals are covered by the Norwegian Medicines Agency whereas personal care products are covered by Norwegian Food Safety Authority. Detailed information (down to gram levels) exists for most pharmaceuticals whereas the consumption of personal care products is far more uncertain.

There are large differences in Ecotoxicological effects of the compounds covered by this screening.

3.2 PPCP as environmental contaminants

PPCPs are a class of new, so-called emerging, contaminants that have raised considerable concern in recent years. PPCPs deserve attention: (i) because of their continuous introduction into the environment via effluents from sewage systems. PPCPs are often described as pseudo-persistent; since their high transformation/removal rates are compensated by their continuous introduction; (ii) in the case of pharmaceuticals they are developed with the intention for performing a biological effect; (iii) PPCPs often have the same type of physio-chemical behaviour as other harmful xenobiotics. Firstly, they are “persistent” to avoid inactivation before they have exerted their curing effect. Secondly, they are hydrophobic to be

able to pass through membranes; and (iv) PPCPs are used by man in rather large quantities (i.e. similar to those of many pesticides) [1].

Pharmaceuticals are involved in one of the greatest environmental chemical mediated catastrophes of our time (the other being methyl mercury in Minamata Bay, Chernobyl, and DDT). Vultures on the Decca peninsula are near extinct due to diclofenac administered to cattle. Diclofenac is nephrotoxic in birds and vultures are exposed to large quantities as they prey on dead cattle. The disappearance of vultures has increased the amount of wild dogs also feeding on dead cattle with a significant increase in rabies both among dogs and people in the area [2].

Selection of compounds
A theoretical study initiated by SFT evaluated and prioritized substances to be included in future environmental monitoring programmes in Norway and Scandinavia. Four groups of high volume chemicals were investigated in this study [3]: 1) Human pharmaceuticals; 2) Aquaculture medicines; 3) Components of personal care products; and 4) Narcotics. The compounds to be included in the screening were selected based on their use, fate, Ecotoxicological effects, and PEC/PNEC ratio (predicted environmental concentration divided by the predicted no effect concentration).

The hospital-use pharmaceuticals were included on different rationale. The antibiotics are still efficient toward most infectious bacteria, and therefore their use should be kept at a low level to postpone (the inevitable) development of resistance. The iodinated x-ray contrast agents are high-volume diagnostic agents that are developed to be inert *in vivo*. That means that they are relatively persistent. These polar compounds are therefore very likely to be detected in environmental samples. The cytostatics were included in the screening due to their toxicity, that is, they are given to patients with cancer to kill cancer cells. These pharmaceuticals are given intravenously, but a portion of the administered dose is excreted un-metabolised through the faeces. Information about the ecotoxicological effects of most cytostatics is scarce, but considered their cytotoxicity to human cancer cells, any presence in the environment should be of some concern. In 1985, 50 tons of antibiotics were used in aquaculture and the development of resistance was an emerging problem [4]. The development of vaccines has led to a decline to almost no use of antibiotics [4]. Anti-parasitic medicines are nowadays used under strict control. Seven anti-parasitic medicines were included to be monitored at two randomly chosen aquaculture plants. Personal care products were included based on the risk assessment conducted in the report [3].

Based on the report [3], SFT suggested a selection of compounds that should be analysed in the Norwegian environment in 2008. The final list of compounds was determined by SFT in collaboration with IVL, NIVA, and NILU. Locations for screening of PPCPs in Tromsø and Oslo were chosen, since there should be a geographical spread in sampling sites chosen for a national screening program. Furthermore, the two places use different waste water treatment technologies and there are also differences in climate.

Below is a brief presentation of these compounds. The structure and CAS number for all the discussed compounds in this report is given in Appendix 1.

3.3 Selected general human pharmaceuticals

The compounds selected from this group are amitriptyline atorvastatin, carbamazepine, morphine, naproxen, paracetamol, propranolol, sertraline, spiramycin, tamoxifen, and warfarin.

Amitriptyline (N06A A09) is a tricyclic antidepressant drug inhibiting serotonin and noradrenalin reuptake almost equally. Amitriptyline has previously been detected in rivers [5] and in STP effluent water [5, 6]. The reported LOEC (*Brachionus calyciflorus*) of amitriptyline is 81 000 ng/L [3]. 292 kg amitriptyline was used in 2006, yielding a PEC/PNEC of 1.05, and it has an estimated bio-concentration factor of as high as 1 226 [3]. The compound degrades slowly in aqueous environments and have the potential to bioaccumulate [7].

Atorvastatin (C10A A05) inhibits HMG-CoA reductase, an enzyme that produces mevalonate, a cholesterol precursor, which lowers the amount of cholesterol produced which in turn lowers the total amount of LDL cholesterol. Atorvastatin has previously been detected in STP effluent water [8, 9]. The reported LOEC (*Lemna gibba*) of atorvastatin is 36 000 ng/L [10]. Statins are high-volume drugs and 864 kg atorvastatin was used in Norway in 2006, yielding a PEC/PNEC of 1.95, but they are extensively metabolised and their environmental effects are largely unknown [3]. Photo degradation is believed to be important for atorvastatin in aquatic environments [10].

The anti-epileptic **carbamazepine** (N03A F01) stabilizes the inactivated state of sodium channels, meaning that fewer of these channels are available to open, making brain cells less excitable (less likely to fire). Only 1-3% is excreted as free carbamazepine, the biologically active 10,11-epoxy-carbamazepine is the major metabolite, glucuronides are minor metabolites [11]. Carbamazepine has been detected in surface waters [5, 8, 11-14], STP influent [5, 12, 15] and effluent water [5, 8, 11-14], and in sludge [16]. The removal efficiency is reported to be 0-55% [8, 12, 17]. The reported LOEC (*Lemna gibba*) of carbamazepine is 25 000 ng/L [18]. In 2006, 3488 kg carbamazepine was used, yielding a PEC/PNEC of 0.21 [3]. Carbamazepine is slowly degraded in the environment ($t_{1/2}$ 82±11 days) [3]. Environmental photo degradation of carbamazepine is important [10, 19] and one transformation product is the very toxic compound acridine [19]. Carbamazepine is prevalent due to poor STP removal [11], with a 50% dissipation time of 82 ± 11 days [10] and is regarded as potentially persistent.

Morphine (N02A A01) is a highly potent opiate analgesic drug, acting directly on the central nervous system to relieve pain, particularly at the synapses of the nucleus accumbens. Morphine has a high potential for addiction; tolerance and both physical and psychological dependence develop rapidly. Heroin (and codeine N02A A59) are partly metabolised to morphine. Morphine has previously been detected in STP effluent water [20]. No ecotoxicological effects of morphine are known, but due to lack of relevant ecotoxicological data, adverse environmental effects from morphine cannot be excluded. The fate of morphine in the environment is unknown.

Naproxen (M01A E02) is a non-steroid anti-inflammatory agent having analgesic and anti-pyretic effect. It acts through inhibition of the enzymes cyclo-oxygenases, which produce prostaglandins. However, the whole mechanism is not fully understood. Naproxen has been

identified in surface waters [5, 8, 14, 21, 22], STP influent [5, 8, 21, 22] and effluent water [5, 8, 13, 14, 21, 22]. For Naproxen, a STP removal efficiency of 40-100% [8], and 67% [23] has been reported. Naproxen has been detected in rainbow trout (*Oncorhynchus mykiss*) exposed to STP effluent water [24]. The reported LOEC (*Ceriodaphnia dubia*) of naproxen is 32 000 ng/L [23]. In 2006, 3814 kg was sold, yielding a PEC/PNEC of 1.7 [3]. Naproxen has no significant bioaccumulation potential (fass.se). Naproxen is susceptible to photo degradation in water [10]. The estimated half-life is 14 days [25].

Paracetamol (N02A A59) works through inhibition of prostaglandin synthesis. Paracetamol has previously been found in surface water [5, 8, 14, 26, 27], STP influent and hospital effluent water [22, 28], STP effluent water [5, 8, 13, 14, 28, 29], and sludge [16, 28]. Paracetamol is reported to be 'efficiently removed' at STP [11], the removal was 98% in a German STP [14] and even a complete removal is reported [8]. The reported LOEC (*Lemna gibba*) of paracetamol is 1 000 000 ng/L [18]. Paracetamol is a high volume drug (173 tons in 2006) with a PEC/PNEC of 5.5. Paracetamol is slowly degraded in the aqueous environment (57% after 28 days), however, its bioaccumulation potential is negligible [3, 7]. .

Propranolol (C07A A05) is a prototype β-blocker that antagonises β1 and β2 adrenoreceptors [30]. Beta-blockers constitute one of the most important families of prescription drugs, and they play a significant pole for the therapy of cardiovascular diseases. Propranolol has previously been measured in surface water [5, 8, 11, 14, 27], STP influent [5, 8, 15], and STP effluent water [5, 8, 11, 14, 15]. The STP removal efficiency was reported to be 96% [11]. The reported LOEC (*Oryzias letipes*) of propranolol is 500 ng/L [8]. In 2006, 367 kg propranolol was consumed, yielding a PEC/PNEC of 21.5 [3]. No information on the fate of propranolol has been found.

Sertraline (N06A B06) is an anti-depressant acting by selectively inhibiting the serotonin re-uptake in CNS. Sertraline has been detected in surface waters [31], STP influent [15] and effluent water [31, 32]. Sertraline is also one of few pharmaceuticals that have been detected in biota [33]. The reported LOEC (*Ceriodaphnia dubia*) of sertraline is 9 000 ng/L [18]. 581 kg was used in 2006, yielding a PEC/PNEC of 3.0 [3]. Sertraline is slowly degraded in the environment [3]. An environmental half life of Sertraline of 4.6 d has been experimentally determined by indirect photolysis (fass.se).

Spiramycin (J01F A02) binds to ribosomes in bacteria, thus inhibiting protein synthesis. Spiramycin has previously been detected in river water [34]. The reported LOEC (*Microcystis aeruginosa*) of spiramycin is 7 000 ng/L [35]. 65 kg was used in 2006, yielding a PEC/PNEC of 3.8 [3]. No information about the environmental fate of spiramycin was found, but the STP removal efficiency of 0% [17], suggests abiotic degradation to be more important than biotic.

Tamoxifen (L02B A01) is a selective estrogen receptor modulator (SERM) that is used in the treatment of breast cancer. Its anti-estrogenic activity is of environmental concern [3]. Tamoxifen has been detected in surface water [8, 27], STP influent and effluent water [8]. A STP removal efficiency of 0% has been reported [8]. Tamoxifen is an important anti-estrogen acting by blocking the estrogen receptor and for environmental risk assessment purposes, tamoxifen citrate has an adverse LOEC concentration 5 600 ng/L [36].

Warfarin (B01A A03) is an anti-coagulant acting by inhibiting the vitamin K-dependent synthesis of biologically active forms of the calcium-dependent clotting factors II, VII, IX and X, as well as the regulatory factors protein C, protein S, and protein Z. Warfarin has

previously been identified in sludge at concentrations up to 92 ng/g d.w. [16]. The reported LOEC (*Pseudokirchneriella subcapitata*) of warfarin is 2 500 000 ng/L (fass.se). Warfarin is also used as a pesticide and its total use is not known. The biodegradation of warfarin was 0% after 28 days (OECD 301D) suggesting a potentially persistency (fass.se). Warfarin hydrolyses very slowly in water with a half-life (pH 7, 25°C) of 16 years [37].

3.4 Selected hospital-use human pharmaceuticals

A hospital-use pharmaceutical is exclusively used in hospitals. This could be due to their toxicity as is the case for cytostatics, some pharmaceuticals require intra venous or intra muscular administration (x-ray contrast agents and certain antibiotics), and some antibiotics are only used for the treatment of severe infections to reduce the possible development of resistance.

3.4.1 β-Lactam antibiotics

The compounds selected from this group are amoxicillin, cefotaxime, cefalotin, meropenem, ofloxacin, penicillin-G, pivmecillinam

Amoxicillin (J01C A04) is a bacteriolytic, β-lactam broad spectrum penicillin antibiotic acting by inhibiting the cross-linkage between the linear peptidoglycan polymer chains that make up a major component of the cell wall of Gram-positive bacteria [18]. The drug is used in aquaculture applications and is also sold as a human pharmaceutical and hence amoxicillin is not exclusively used in hospitals [3]. Amoxicillin has not previously been detected in environmental samples. The reported LOEC (*Pseudokirchneriella subcapitata*) of amoxicillin is 2 200 ng/L [38]. 1880 kg of amoxicillin was sold in Norway in 2006 yielding a PEC/PNEC of 149 [3]. Amoxicillin is slowly degraded in the environment, with a hydrolytic half-life of 50-113 days at pH 7 (OECD 111) and a photolytic half-life of 1.13 days at pH 7.5 [3].

Cefotaxime (J01D D01) is administered intravenously and is a 3rd generation cephalosporin that inhibits bacterial cell wall synthesis by binding to penicillin-binding proteins, which in turn inhibits the final transpeptidation step of peptidoglycan synthesis in bacterial cell walls. Cefotaxime has not previously been detected in environmental samples. Cefotaxime has a reported toxicity to Zebra fish *Danio rerio* (EC_{50} 96 h) of > 500 000 000 ng/L [3]. Ash et al carried out a study on water samples taken from streams in USA and found evidence of bacterial resistance to e.g. cefotaxime [39]. Cefotaxime is potentially persistent with a 13% degradation in 28 days, but the substance is light sensitive [3].

Cefalotin (J01D B03) is administered intravenously and a 1st generation cephalosporin that inhibits the cell wall synthesis in bacteria. Cefalotin has not previously been detected in environmental samples. No data on the Ecotoxicological effects of cefalotin has been found, and the information regarding the environmental fate of cefalotin is scarce.

Meropenem (J01D H02) is administered intravenously and is a carbapenem that inhibits bacterial wall synthesis like other beta-lactam antibiotics. Meropenem is a typical hospital antibacterial agent. Meropenem has not previously been detected in environmental samples. Meropenem has a reported EC_{50} (48 h) of >900 000 000 ng/L to *Daphnia magna* [3]. . Meropenem is not rapidly biologically degraded, but it is prone to undergo hydrolysis with reported half lives of 63 h (pH 7) and 12 min (pH 9). Its potential for bioaccumulation is low [3].

Ofloxacin (J01M A01) is administered both per oral and intra venous. It is a fluoroquinolone antibiotic and acts by inhibiting the enzyme DNA-gyrase [18]. Ofloxacin has previously been detected in river water [34], STP influent and STP effluent water [12]. A STP removal rate of 57% was reported for ofloxacin [17]. The reported LOEC (*Synechococcus leopolensis*) of ofloxacin is 5 000 ng/L [18]. In 2006, 28 kg ofloxacin was used, yielding a PEC/PNEC of 0.6 [3]. Fluoroquinolones are known to be very persistent in the environment [3]. Ofloxacin strongly adsorbs to soil and is highly active in hospital wastewaters [11, 40]. The medicine shows no biodegradation, but the substance is light sensitive [10].

Benzyl penicillin, commonly known as **penicillin G** (J01C E01), is administered intravenously and is a beta-lactamase sensitive penicillin that acts by inhibiting synthesis of cell walls in bacteria. Penicillin G has not previously been detected in environmental samples. The reported LOEC (*Microcystis aeruginosa*) of penicillin G is 6 000 ng/L [35]. 1588 kg benzyl penicillin was sold in 2006 yielding a PEC/PNEC of 77 [3]. Penicillin G is reported to be unstable due to hydrolysis and photolysis [35].

Pivmecillinam (J01C A08) is bactericide broad spectrum penicillin administered per orally that act by inhibition of cell wall synthesis, but in a different way than other penicillins. Pivmecillinam has not been detected in environmental samples, no ecotoxicological data are currently available. In 2006, 1487 kg pivmecillinam was used, yielding a PEC/PNEC of 0.73 [3]. The fate of pivmecillinam in the environment is unknown.

3.4.2 X-ray contrast agents

The compounds selected from this group are iodixanol, iohexol and iopromide.
The iodinated pharmaceuticals **iodixanol** (V08A B09), **iohexol** (V08A B02), and **iopromide** (V08A B05) are used in diagnostics and not for the treatment of any diseases. Their mode of action is to block x-rays (due to the high electron density of the iodine atom) as they pass through the body. The three compounds have the same mechanism of action and presumably very similar physio-chemical properties and they are therefore discussed together. Iopromide has previously been detected in STP effluent water with no effective removal in the STPs [11, 41, 42]. No reports on the detection of iohexol and iodixanol in the environment were found. The toxicity of the metabolites of iopromide are unknown [11]. Iopromide is toxic towards a (unspecified) cyanobacterium with an EC_{50} of 68 000 000 ng/L [11]. It has also been tested to the invertebrate *Daphnia magna*, yielding an EC_{50} of >1 000 000 000 ng/L [18]. No reports on the Ecotoxicological effects of iohexol and iodixanol were found. It is estimated that 100-200 tons iodinated contrast media are annually consumed in Europe [3]. Iopromide is very resistant to biodegradation and extremely persistent [11]. No reports on the fate of iohexol and iodixanol in the environment were found.

3.4.3 Cytostatics

The compounds selected from this group are bortezomib, docetaxel, doxorubicin (and doxorubicinol) irinotecan, paclitaxel (and 6-OH-paclitaxel).

Bortezomib (L01X X32) acts by binding of the boron atom to the catalytic site of the 26S proteasome. Bortezomib has not been detected in environmental samples. The reported LOEC (*Scenedesmus subspicatus*) of bortezomib is 100 000 ng/L [3]. No information is available on degradation and bioaccumulation of bortezomib.

Docetaxel (L01C D02) acts through de-polymerization of microtubule, hence inhibiting cell division. Docetaxel has not previously been detected in environmental samples. An EC_{50} (48 h) of 3 700 000 ng/L for *Daphnia magna* and the EC_{50} (72 h) is 545 000 ng/L for the algae *Scenedesmus subspicatus* are reported. Docetaxel is slowly degraded with a hydrolytic half-life at pH 7 of 28 days. Bioaccumulation of docetaxel cannot be excluded [3].

Doxorubicin (L01D B01) and its active metabolite doxorubicinol presumably act by interfering with DNA base pairing and hence inhibit replication. Doxorubicin has previously been detected at 500 ng/L in hospital effluent water [43, 44]. Doxorubicin is toxic to *Daphnia magna*, with a reported toxic concentration (EC_{50}) of 9 900 000 ng/L [3]. No information is available regarding the degradation of doxorubicin in the environment [3], but it should be prone to photo degradation, due to its intense beautiful red colour. No data are available on the degradation and bioaccumulation on doxorubicin and doxorubicinol.

Irinotecan (L01X X19) is a derivative of camptothecin and inhibits DNA-topoisomerase I, an enzyme involved in DNA-replication. Irinotecan has not previously been detected in the environment, and no ecotoxicological data are available for the compound.
Irinotecan is extensively used, but the fate of irinotecan in the environment is not known.

Paclitaxel (L01C D01) and its metabolite 6-OH-paclitaxel act by inhibiting the de-polymerization of microtubuli. Paclitaxel and 6-OH-paclitaxel have not been found in environmental samples. For Paclitaxel, a NOEC of 740 000 ng/L is reported for *Daphnia magna* [3]. Paclitaxel is readily degraded in the environment [3]. Paclitaxel has a log Kow of 3.5 (pH 7), however, the bioaccumulation potential to organisms is low based on metabolism and biodegradation data. Paclitaxel is readily biodegraded as it exhibited 68.1% mineralization to $^{14}CO_2$ in the first 14 days of a biodegradation study [3].

3.5 Selected aquaculture pharmaceuticals

The compounds selected from this group are cypermethrin, deltamethrin, emamectin, fenbendazole, flumequine, oxolinic acid, and praziquantel.

Cypermethrin (no ATC code) and **deltamethrin** (QP53A C11) are anti-parasitic agents, and act by altering sodium channels in nerve cells, causing depolarization, paralysis and death.
Cypermethrin and deltamethrin have previously been detected in river sediments and in river water [45]. The pyrethroid insecticides have been reported to be toxic to *Hyalella azteca* [45] and *Vibrio fischeri* (EC_{50} of >39 900 000 ng/L for deltamethrin) [46]. In 2006, 57 kg deltamethrin was used, yielding a PEC/PNEC of 67 [3]. In the same year, 49 kg cypermethrin was used, yielding a PEC/PNEC of 2.1 [3]. The fate of cypermethrin and deltamethrin in the environment is scarcely described, but it is suggestive that the compounds will adsorb to solids. .
Emamectin (QP54A A06) is an anti-parasitic agent (used on salmon) and acts through binding of invertebrate glutamate regulated ion channels. Emamectin has not previously been detected in environmental samples. The reported LOEC (*Vibrio fischeri*) of emamectin is 6 300 000 ng/L [46]. In 2006, 60 kg emamectin was used, yielding a PEC/PNEC of 191 [3]. The fate emamectin in the environment is not known.

Fenbendazole (QP52A C13) is a broad spectrum benzimidazole anti-helminitic agent, inhibiting carbohydrate metabolism in nematodes and is neurotoxic to cestodes. Fenbendazole

has not previously been detected in environmental samples. An EC_{50}-48 h of 16 500 ng/L of fenbendazole to *Daphnia magna* is reported [47]. In 2006, 1 038 kg was sold in Norway, yielding a PEC/PNEC of 0.70 [3]. The fate of fenbendazole in the environment is unknown.

Flumequine is a quinolone, acting by inhibiting DNA gyrase, and is a broad spectrum antibiotic often used in veterinarian medicine. Flumequine has not previously been detected in environmental samples. The reported LOEC (*Vibrio fischeri*) of flumequine is 198 000 [46]. In 2006, 7 kg flumequine was sold, yielding a PEC/PNEC of 0.003 [3]. Information about the fate of flumequine in the environment is scarce.

Oxolinic acid (QJ01M B91) is a quinolone acting by inhibiting DNA gyrase. Oxolinic acid has previously been detected in shrimp [48]. The reported LOEC (*Vibrio fischeri*) of oxolinic acid is 200 000 [46]. In 2006, 1119 kg oxolinic acid was used, yielding a PEC/PNEC of 1.9 [3]. No information about the fate of oxolinic acid in the environment is available.

Praziquantel (QP52A A01) acts by inducing damage to the parasite's integumentary system, leading to paralysis. Praziquantel has not previously been detected in environmental samples. Praziquantel has a NOEL for vertebrates at 20 000 000 ng/kg/day [49]. Praziquantel was determined to have a NOEC of >1 000 000 000 ng/kg dung to the larvae of the dung beetle *Aphodius constans* [49]. In 2006, 145 kg praziquantel was used, yielding a PEC/PNEC of 3.7 [3]. The fate of praziquantel in the environment is not known.

3.6 Selected personal care products

The compounds selected from this group are avobenzone, butyl paraben, cetrimonium salt, cocoamidopropyl betaine (CAPB), diethyl phthalate (DEP), ethylene-diaminotetraacetic acid (EDTA), sodium dodecyl sulphate (SDS), and sodium laureth sulphate (SDSEO).

Avobenzone is also known as butyl methoxydibenzoylmethane, BMDBM and Eusolex 9020 [50]. Avobenzone is the most frequently used UV filter and is only currently registered UV filter with a strong absorbance in the UV-A region [51]. Avobenzone has previously been found in swimming pools and in surface water [52-54]. Avobenzone showed no endocrine disrupting activity when tested for estrogenic activity (MCF-7 cells) or anti-androgenic activity (MDA-kb2 cells) [55]. Avobenzone showed no estrogenic activity on rainbow trout estrogenic receptor (rtER) and human ER (hER) [56]. Avobenzone has a bio-concentration factor of 85 and is not readily degraded in the environment and potentially bioaccumable [3]. Avobenzone degrades in sunlight (www.smartskincare.com).

Butyl paraben is a preservative agent used in personal care products. Due to suspected adverse effects and a weak link with breast cancer, the use of parabens is declining. Butyl paraben has previously been detected in STP influent [5, 57] and effluent water [57, 58], and in sludge [57]. A removal efficiency of 96% for butyl paraben in a WWTP was observed [58]. Parabens are weak estrogens [58, 59]. The anti-androgenergic effect of butyl paraben was investigated [60], and it inhibited testosterone induced transcriptional activity by 19% at 1 940 000 ng/L. A PEC/PNEC of 0.002 has been calculated for butyl paraben [3]. Butyl paraben has a bio-concentration factor of 110, and parabens are not expected to undergo hydrolysis in the environment [3]. Butyl paraben is stable against photo degradation, but is readily biodegradable with half-times varying between 9.5 and 16 h [58].

Cetrimonium salts belong to a group of compounds commonly known as alkyltrimethylammonium chlorides (ATAC), which is widely used as surfactant, bactericide, and algaecide [61]. An estimated use of cetrimonium salts of 24 000 kg (2006) yields a PEC/PNEC of 360 [3]. Cetrimonium has previously been detected in sludge and river sediments [61]. The reported LOEC (*Microcystis* sp.) of cetrimonium is 25 000 ng/L [62]. The fate of cetrimonium in the environment is not known.

Cocoamidopropyl betaine (CAPB) is a quaternary ammonium compound (QAC), an economically important class of industrial chemicals. Because of their physical and chemical properties they are used as disinfectants, surfactants, anti-electrostatics (e.g. in shampoo), and phase transfer catalysts. QAC belong to the group cationic surfactants, hence they are located at the phase boundary between the organic and the water phase. They therefore have the capacity to attach themselves onto specific sites of the bacterial cell membrane and block the up-take of nutrients into the cell and prevent the excretion of waste products, which accumulate within its structure [61]. Cocoamidopropyl betaine has not previously been monitored in the environment. The reported LOEC (*Skeletonema costatum*) of cocoamidopropyl betaine is 260 000 [63]. In 2006, an estimated release of 236 400 kg CAPB yields a PEC/PNEC of 1773 [3]. The alkyl chain may undergo β- or ω-oxidation.

Diethyl phthalate (**DEP**) is a plasticiser, i.e., a substance added to plastics to increase their flexibility. Phthalates are chiefly used to soften polyvinyl chloride. Phthalates are being phased out of many products in the United States and European Union over health concerns. DEP has previously been detected in river waters [64, 65], sediment [66], and all other environmental compartments [67]. Phthalates have been shown to be endocrine disruptors (weak estrogen mimics) [68]. In a study from India, infertile men had significantly higher DEP concentration in their semen than fertile men [69]. Estrogen mimicking activity was observed in *Cyprinus carpio* at concentrations of 96 000 ng/L, which is 500 times lower than the LC_{50} of the same species [67]. An estimated use of 15 000 kg (2006) yields a PEC/PNEC of 0.62 [3]. The aqueous hydrolysis half-life of DEP is 8.8 yr, whereas the atmospheric half life is 1.8-18 days [70]. In soil, 90% of inoculated DEP was degraded within a week [70].

EDTA is used as a chelating agent due to its ability to "sequester" di- and tri-cationic metal ions. This is very useful in areas with hard water, as Ca^{2+} and Mg^{2+} ions are efficiently inactivated. EDTA has been detected in surface waters [72]. One possible mechanism for EDTA Ecotoxicological effects is through enhanced uptake of undesired metal cations. A LD_{50} of 24 000 000 ng/L was reported for bluegill (*Lepomis macrochirus*) [73]. The global production of EDTA was estimated roughly as 100 000 tons in 2001 [71]. The greatest consumer in Scandinavian area is the pulp and paper industry. EDTA is used as a stabilizer in the hydrogen peroxide bleach processes. An estimated release of 14 000 kg (2006) yields a PEC/PNEC of 0.23 [3]. EDTA is only slowly biodegradable, and therefore is rather persistent in the environment [71, 74]. An important sink for EDTA in the environment is photo degradation but is only valid for the Fe-EDTA complex [72, 75-77]. EDTA may be degraded under special conditions in the activated sludge in STP [78, 79].

Sodium dodecyl sulfate (**SDS**), or sodium lauryl sulfate, is a detergent used in soaps and shampoos as it is efficient for sebum removal (along with dead skin cells, dirt, and the bacteria living on it) [80]. SDS has not previously been analysed in environmental samples. The reported LOEC (*Skeletonema costatum*) of SDS is 360 000 ng/L [63]. An estimated use of 1 990 000 kg (2006) yields a PEC/PNEC of 15 [3]. SDS may undergo β-oxidation mediated by *Pseudomonas* sp. [81, 82].

Sodium laureth sulfate (**SDSEO**) is a detergent used in soaps and shampoos as it is efficient for sebum removal (along with dead skin cells, dirt, and the bacteria living on it) [80]. It has a better water solubility than SDS at low temperatures and is therefore the preferred detergent in soaps and shampoos. Sodium laureth sulfate has not been detected in other environmental samples. The reported LOEC (*Skeletonema costatum*) of SDSEO is 370 000 ng/L [63]. An estimated release of 3 752 400 kg (2006) yields a PEC/PNEC of 563 [3]. The detergent sodium laurylether sulfate may undergo ω-oxidation [83].

3.7 Samples and sampling

Following an agreement with SFT, it was decided that the pharmaceuticals should be analysed in samples taken from two locations in Norway, Oslo and Tromsø. In the Oslo area, samples were collected from Ullevål hospital (hospital effluent water), VEAS STP (sewage treatment plant): effluent water and sludge, Inner Oslofjord: receiving water, sediment and blue mussel from Ramton and Gåsøya. In Tromsø, samples were taken from the University Hospital of Northern Norway: hospital effluent water; Breivika STP: effluent water and sludge; Tromsøsund: receiving water, sediment and blue mussel.

Two fish farms were also included to analyse the content of the aquaculture medicines listed above in water and sediment samples taken in close proximity from the farms.

Details on the sampling procedures and chemical analysis are given in Chapter 4. The results are given in Chapter 5 where the results also are discussed. The conclusions of the study are presented in Chapter 6.

4. Materials and methods

4.1 Description of sampling sites

Four locations were selected for the collection of samples to address the potential release and accumulation of pharmaceuticals in the marine environment:

1. The inner Oslofjord in the vicinity of Norway's largest wastewater treatment plant (Vestfjorden avløpsselskap, VEAS) was selected based on the volume of hospital wastewaters reaching the treatment plant and the advanced treatment applied here. Being one of the major hospitals with discharge to VEAS, and treating patients with a broad spectrum of somatic illnesses, including cancer and psychiatric patients, the main effluent from Ullevål University hospital was included in the sampling campaign. VEAS discharges at ca 50 m depth on the Slemmestad.
2. The University hospital Nord-Norge (UNN) in Tromsø has discharge to the simple mechanical treatment plant Breivika, which has its discharge to Tromsøsund. Most of the prioritized antineoplastic pharmaceuticals are used in treatment at UNN and the UNN discharge constitutes ca 1/3 of the total discharge to Breivika.
3. Fish farm No. 1 in Bømlafjord for addressing pharmaceuticals used in aquaculture.
4. Fish farm No. 2 in Romsdalsfjord for addressing pharmaceuticals used in aquaculture.

An additional criterion for selection of locations was that they should be in relative close proximity of an office of one of the participating Institutes or situated along the pre-planned route of an ongoing sampling campaign.

A total of 64 samples were analysed and included samples from hospital effluents (8), water effluents (8) and final sludge effluents (4) from waste water treatment plants, seawater (20), sediment (16) and blue mussel (8). In addition to this blank samples (4) were collected. A more detailed description of the each station is given below and summarized in Table 1 and shown on maps in Figures 1-5. Figure 1 shows the main sampling locations, whereas the Figures 2-5 give a detailed view of the different sampling sites.

Each sample was further divided in 3 to 6 sub-samples depending on which analysis were to be performed.

Table 1: Summary of samples; main area, sampling station and GPS coordinates, sample category and sample matrix.

Area	Station	Category	Matrix
Inner Oslofjord	Ullevål hospital	hospital effluent	water
Inner Oslofjord	VEAS STP	blank	water
Inner Oslofjord	VEAS STP	STP effluent	water
Inner Oslofjord	VEAS STP	STP effluent	sludge
Inner Oslofjord	Slemmestad bank	blank	water
Inner Oslofjord	Slemmestad bank	receiving	water
Inner Oslofjord	Slemmestad bank	receiving	water
Inner Oslofjord	Slemmestad bank	receiving	water
Inner Oslofjord	Slemmestad bank	receiving	water
Inner Oslofjord	Slemmestad bank	receiving	water
Inner Oslofjord	Slemmestad bank	receiving	sediment
Inner Oslofjord	Slemmestad bank	receiving	sediment
Inner Oslofjord	Slemmestad bank	receiving	sediment
Inner Oslofjord	Gåsøya	receiving	blue mussels
Inner Oslofjord	Ramton	receiving	blue mussels
Tromsøsund	UNN hospital/ Breivika STP	hospital effluent	water
Tromsøsund	Breivika STP	blank	water
Tromsøsund	Breivika STP	STP effluent	water
Tromsøsund	Breivika STP	STP effluent	sediment
Tromsøsund	Breivika STP	STP effluent	sediment
Tromsøsund	Tromsøy strait	blank	water
Tromsøsund	Tromsøy strait	receiving	water
Tromsøsund	Tromsøy strait	receiving	water
Tromsøsund	Tromsøy strait	receiving	water
Tromsøsund	Tromsøy strait	receiving	water
Tromsøsund	Tromsøy strait	receiving	water
Tromsøsund	Tromsøy strait	receiving	sediment
Tromsøsund	Tromsøy strait	receiving	sediment
Tromsøsund	Tromsøy strait	receiving	sediment
Tromsøsund	Tromsøy strait	receiving	blue mussels
Tromsøsund	Tromsøy strait	receiving	blue mussels
Bømlafjord	Fish farm 1	blank	water
Bømlafjord	Fish farm 1	receiving	water
Bømlafjord	Fish farm 1	receiving	water
Bømlafjord	Fish farm 1	receiving	water
Bømlafjord	Fish farm 1	receiving	water
Bømlafjord	Fish farm 1	receiving	water
Bømlafjord	Fish farm 1	receiving	sediment
Bømlafjord	Fish farm 1	receiving	sediment
Bømlafjord	Fish farm 1	receiving	sediment
Bømlafjord	Fish farm 1	receiving	sediment
Bømlafjord	Fish farm 1	receiving	sediment
Bømlafjord	Fish farm 1	receiving	blue mussels
Bømlafjord	Fish farm 1	receiving	blue mussels

Table 1 (continued): Summary of samples; main area, sampling station and GPS coordinates, sample category and sample matrix.

Area	Station	Category	Matrix
Romsdalsfjord	Fish farm 2	receiving	water
Romsdalsfjord	Fish farm 2	receiving	water
Romsdalsfjord	Fish farm 2	receiving	water
Romsdalsfjord	Fish farm 2	receiving	water
Romsdalsfjord	Fish farm 2	receiving	water
Romsdalsfjord	Fish farm 2	receiving	sediment
Romsdalsfjord	Fish farm 2	receiving	sediment
Romsdalsfjord	Fish farm 2	receiving	sediment
Romsdalsfjord	Fish farm 2	receiving	sediment
Romsdalsfjord	Fish farm 2	receiving	sediment
Romsdalsfjord	Fish farm 2	receiving	blue mussels
Romsdalsfjord	Fish farm 2	receiving	blue mussels

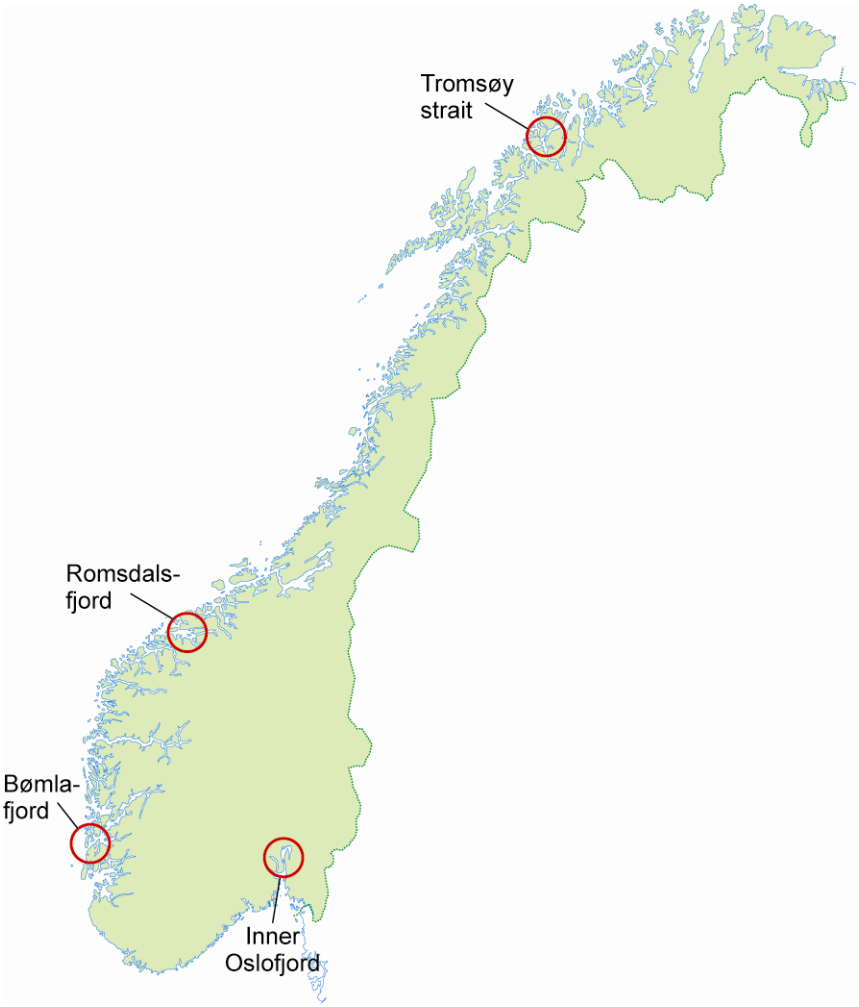

Figure 1: Map showing the main sampling locations for the screening program

4.1.1 Hospitals and their wastewater discharges

Ullevål University hospital is one of the largest hospitals in Oslo having ca. 45 000 hospitalisations and ca. 400 000 patient consultations per year within a broad spectra of somatic illnesses, including cancer and psychiatric patients. The hospital has untreated discharge to the domestic sewage system which ends up at Vestfjorden Avløpsselskap (VEAS).

The University hospital Nord-Norge (UNN) is a university hospital within psychiatry and somatic units. The hospital offers specialist care for the whole of the north of Norway. Of the antineoplastic pharmaceuticals included in the prioritised list all are used in treatment at UNN. The hospital discharges directly to domestic sewage and constitute on average ca. 1/3 of the influent to Breivika wastewater treatment plant.

4.1.2 Wastewater treatment plants and their discharges

Vestfjorden Avløpsselskap (VEAS) is the largest wastewater treatment plant in Norway with discharge of domestic and industrial wastewater from a population of 440 000 in Oslo, *Bærum, Asker, Røyken and Nesodden (Figure 4, ●). The plant receives yearly 100 - 110* million m^3 of wastewater that is treated mechanically, chemically and biologically (post-denitrification). The sludge is treated by anaerobic digestion and drying ending in the product "VEAS-jord", ca 25 000 tons per year with a dry content of 51 - 59%. The treatment plant receives the wastewater from all the major hospitals in the area, including Ullevål University hospital. VEAS discharges the treated water at a depth of ca. 50 m in the Oslofjord.

Breivika wastewater treatment plant in Tromsø municipality (Figure 5, ●) receives domestic wastewater from a total of 2 850 households and the University hospital Nord-Norge (UNN). The wastewater is treated by simple screening (0.35 mm mesh size) and the plant has a capacity of 18 700 person equivalents. The removed sludge dewatered in a screw press and sent to Balsfjord municipality (Stormoen) for windrow composting. The treated wastewater is discharged at a depth of 30 m and ca. 300 m out into the Tromsø strait.

4.1.3 Fish farms

The two fish farms to be included in the screening were selected by the Norwegian Pollution Control Authority (SFT) in collaboration with the Norwegian Food Safety Authority (Mattilsynet) from a list of Norwegian fish farms retrieved from the Directorate of Fisheries (Fiskeridirektoratet). The main criterion for the selection was that they were using aquaculture medicines just before sampling. Fish farm 1 (a salmon farm) used emamectin benzoate which started 30.06.2008 and ended 06.07.2008 and deltamethrin which started 07.01.2008 and ended 31.12.2008. Fish farm 2 (a cod farm) used oxolinic acid starting on 11.07.08 and finishing treatment on 21.07.08 (information from Mattilsynet).

Fish farm No. 1 is located in the Bømlafjord area (Figure 2). At the sampling time there were three fish net cages and the outer one was not in use.

Fish farm 2 is located in Romsdalsfjord. A satellite photo of the farm is shown in Figure 3. At the sampling time there were three fish net cages and only the inner one was in use.

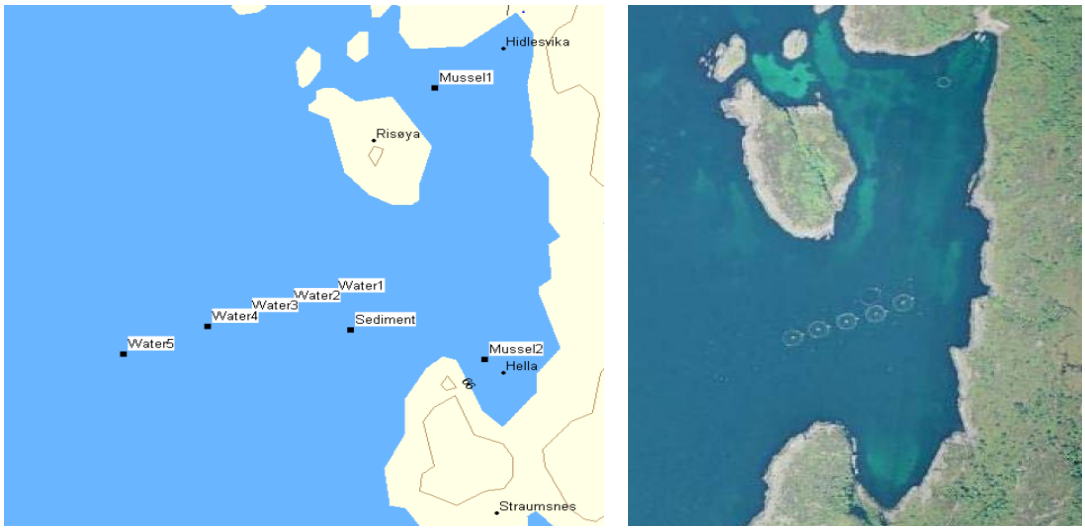

Figure 2: Satellite photo (right) of the fish farm 1 in the Bømlafjord area (http://kart.sesam.no/) and a map (left) showing the sampling stations in the same area. At the time of sampling there were three fish net cages and the outer one was not in use. Mussel station 1 was located at the empty fish cage north of the others.

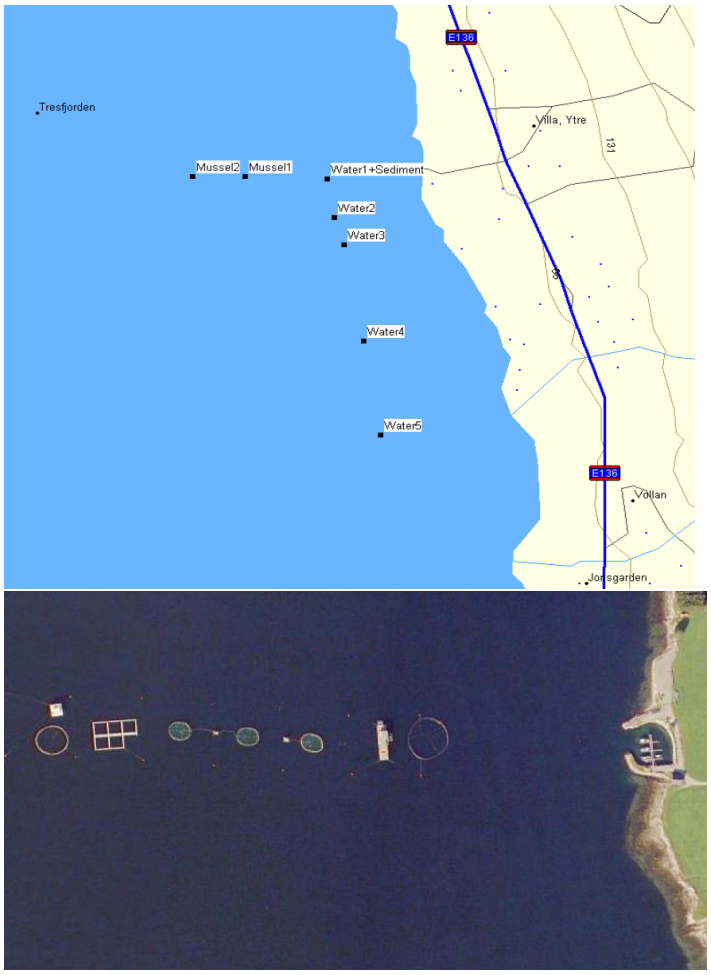

Figure 3: Satellite photo (below) of fish farm 2 in the Romsdalsfjord area. (http://kart.sesam.no/) and a map (above) showing the sampling stations in the same area. At the time of sampling there were three fish net cages and only the inner one was in use. Mussel station 1 and 2 were located at the third fish cage.

4.1.4 Marine sampling stations

The different marine sampling stations are shown in the following Figure 4 and Figure 5.

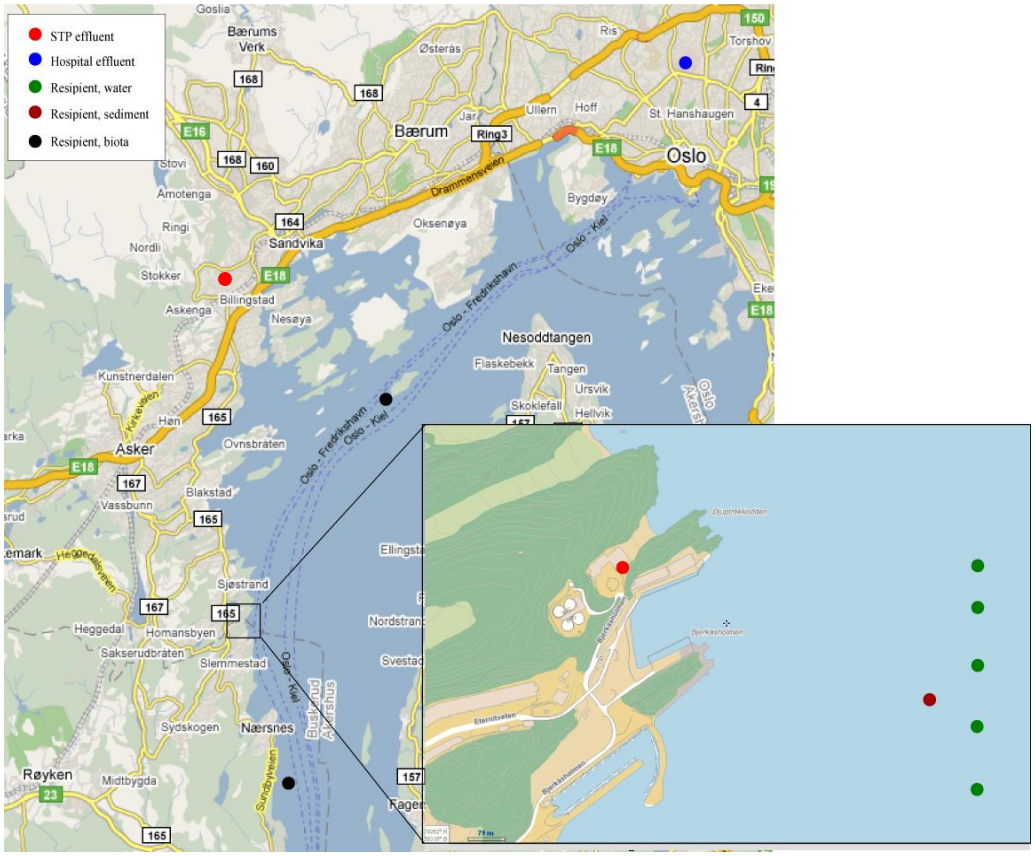

Figure 4: Map of sampling stations in the inner Oslofjord area in the vicinity of the effluent pipeline from VEAS. VEAS is marked by a red dot. Blue mussel stations north at Gåsøya and south at Ramton are not shown on the map.

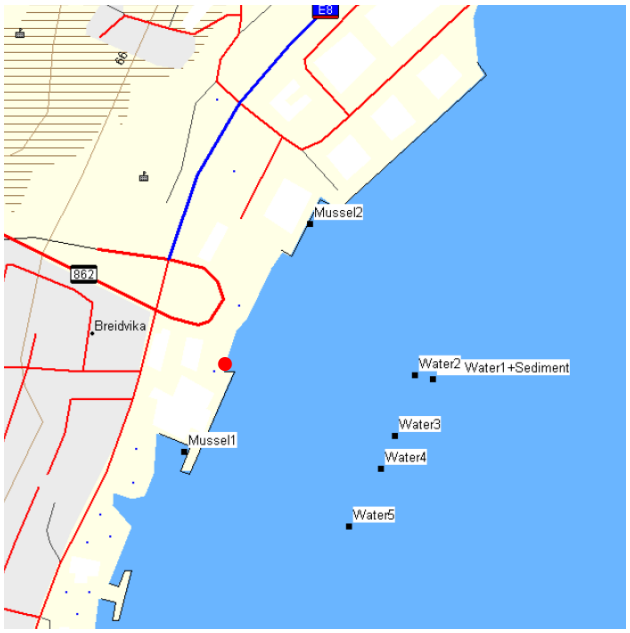

Figure 5: Map of sampling stations in the Tromsøsund area in the vicinity of the effluent pipeline from Breivika STP. The treatment plant is marked by a red dot.

4.2 Sampling and sample treatment

4.2.1 Sampling bottles

The collected samples were divided and transferred to different pre-prepared bottles depending on which analyses were to be performed. A description of sample bottles and method of preservation is shown in Table 2.

Table 2: Containers for sample collection

Receiving samples	Containers	Preservative
Seawater		
NIVA-1	Silanised amber glass bottles 2.5 litre	
NIVA-2	Silanised amber glass bottles 2.5 litre	
NILU-1	Brown plastic 2.5 litre	CDTA
NILU-2	Silanised amber glass bottles 2.5 litre	
NILU-3	PP 2.5 litre	2 % BSA
IVL-1	Brown plastic 2,5 litre	pH 2
IVL-2	Baked (500 °C) amber glass 2.5 litre	pH 2
Sediment and sludge		
NIVA-1	Plastic container 100 ml	
NILU-1	Plastic container 100 ml	
NILU-3	Plastic container 100 ml	
IVL-1	Plastic container 100 ml	
Blue mussels		
NIVA-1	Baked (500 °C) glass	
NILU-1	Baked (500 °C) glass	
IVL-1	Baked (500 °C) glass	

4.3 Hospital wastewater sampling

4.3.1 Sampling at Ullevål University hospital

Four 24 hour composite samples were collected from the combined sewage from the whole hospital area between September 3rd and 12th, 2008 (Table 3).

Table 3: Details regarding sampling of wastewater effluent samples from Ullevål University hospital.

Station	Sample type	Period	Flow (m^3/d)	Sampling equipment	Analysis
Ullevål hospital effluent	water	03.09 09:30 - 04.09 10:00	6181	Isco 6712, Isco 2150	NILU-1, 2, and 3
Ullevål hospital effluent	water	08.09 09:10 - 09.09 10:25	4731	Isco 6712, Isco 2150	NILU-1, 2, and 3
Ullevål hospital effluent	water	09.09 10:30 - 10.09 09:30	3442	Isco 6712, Isco 2150	NILU-1, 2, and 3
Ullevål hospital effluent	water	11.09 09:50 - 12.09 09:10	3238	Isco 6712, Isco 2150	NILU-1, 2, and 3

Since the wastewater is severely influenced by surface runoff, as documented by flow measurements during heavy rain prior to the sampling period, sampling was conducted during no, or limited, precipitation. The water and sanitation office of the municipality (VAV) supplied an automatic composite sampler (Isco 6712) for collecting a sub-sample every 20 min from a manhole on the camp site of the hospital. The wastewater flow was monitored in

the same period by an Isco 2150 Area Velocity Module installed by NIVA 50 m downstream of the sampler. Personnel at VAV prepared and installed the automatic sampler and conducted the sampling, the latter in accordance with a protocol described by NIVA.

4.3.2 Sampling of the UNN effluent

Since there were no mobile automatic samplers available in Tromsø, and since the wastewater flow from UNN constitutes on average a third of the influent to Breivika treatment plant, the automatic sampling equipment mounted to monitor this influent at the treatment plant was used to collect 4 × 24 hour flow-proportional composite samples of UNN wastewater effluent (Table 4). Sampling and sample handling was conducted by personnel at the treatment plant in accordance with a protocol described by NIVA. The collected composite sample was mixed well before being split by transferring to the different pre-prepared sampling bottles. Each bottle was labelled with location and sampling period. The bottles were packed securely together with cooling elements and transported as soon as possible to the appropriate receiving laboratories followed by an e-mail to the receivers. The bottles for NILU-1 and NILU-3 were covered by aluminium foil to protect them from sun light. All handling of samples were done using powder-free nitrile gloves. The automatic sampler was cleaned thoroughly between each new composite sample.

Table 4: Details regarding sampling of wastewater effluent samples from UNN.

Station	Sample type	Period	Flow (m^3/d)	Sampling equipment	Analysis
UNN effluent	water	25.08 07:30 - 26.08 07:30	6294	Water PSW 2000	NILU-1, 2, and 3
UNN effluent	water	26.08 07:45 - 27.08 07:45	6269	Water PSW 2000	NILU-1, 2, and 3
UNN effluent	water	27.08 08:00 - 28.08 08:00	6429	Water PSW 2000	NILU-1, 2, and 3
UNN effluent	water	29.08 08:15- 30.08 08:15	6591	Water PSW 2000	NILU-1, 2, and 3

4.4 Wastewater treatment plant sampling

4.4.1 Sampling at VEAS

Four 24 hour flow-proportional composite samples were collected at VEAS using the same sampling equipment that is used to do the daily effluent sampling at the treatment plant (Table 5). Collection of wastewater samples and sample handling was conducted by personnel at VEAS in accordance with a protocol described by NIVA and briefly outlined above for sampling of effluent from UNN. In addition, two sludge samples were collected during the wastewater sampling period. The sludge samples were collected from the final sludge and transferred to the sample containers by a clean spoon or similar. The samples were marked, packed and sent away as described for the water samples. During the sampling campaign a blank sample bottle with deionised water was stored open in the same environment as the composite sample container during the 24 hour sampling period. The bottle was protected from direct drop contamination.

Table 5: Details regarding sampling of wastewater effluent and sludge samples from VEAS. The water sampler was from Contronic Development AB, Sweden.

Station	Sample type	Period	Flow (m3/d)	Sampling equipment	Analysis
VEAS effluent	water	02.09 08 - 03.09 08	260000	Water sampler PSW 2000	NIVA-1, NILU-1, 2, and 3, IVL-1 and 2
VEAS effluent	water	03.09 08 - 04.09 08	332000	Water sampler PSW 2000	NIVA-1, NILU-1, 2, and 3, IVL-1 and 2
VEAS effluent	water	11.09 08 - 12.09 08	343000	Water sampler PSW 2000	NIVA-1, NILU-1, 2, and 3, IVL-1 and 2
VEAS effluent	water	15.09 08 - 16.09 08	247000	Water sampler PSW 2000	NIVA-1, NILU-1, 2, and 3, IVL-1 and 2
Blank	water	15.09 08 - 16.09 08		Water sampler PSW 2000	NIVA-1, NILU-1, 2, and 3, IVL-1 and 2
VEAS effluent	sludge	04.09			NIVA-1, NILU-1, NILU-3, IVL-1
VEAS effluent	sludge	04.09			NIVA-1, NILU-1, NILU-3, IVL-1

4.4.2 Sampling at Breivika WWTP

Four 24 hour flow-proportional composite samples were collected at Breivika WWTP using the same sampling equipment that is used to do the regular effluent sampling at the treatment plant. In addition two final stage sludge samples were collected (Table 6). Sampling and sample handling was conducted by personnel at the treatment plant in accordance with a protocol described by NIVA and briefly outlined above for sampling of effluent from UNN and VEAS.

Table 6: Details regarding sampling of wastewater effluent and sludge samples from Breivika WWTP.

Station	Sample type	Period	Flow	Sampling equipment	Analysis
Breivika effluent	water	25.08 07:30 26.08 07:30	6294 m^3/d	Water PSW 2000	NIVA-1, NILU-1, 2, and 3, IVL-1 and 2
Breivika effluent	water	26.08 07:45 27.08 07:45	6269 m^3/d	Water PSW 2000	NIVA-1, NILU-1, 2, and 3, IVL-1 and 2
Breivika effluent	water	27.08 08:00 28.08 08:00	6429 m^3/d	Water PSW 2000	NIVA-1, NILU-1, 2, and 3, IVL-1 and 2
Breivika effluent	water	29.08 08:15 30.08 08:15	6591 m^3/d	Water PSW 2000	NIVA-1, NILU-1, 2, and 3, IVL-1 and 2
Blank		25.08 07:30 - 26.08 07:30	-	Water PSW 2000	
Breivika effluent	sludge	27.07	0.25 ton/d	Water PSW 2000	NIVA-1, NILU-1, NILU-3, IVL-1
Breivika effluent	sludge	27.07	0.25 ton/d	Water PSW 2000	NIVA-1, NILU-1, NILU-3, IVL-1

4.5 Sampling in the receiving waters

The water samples were collected by a Niskin water sampler (5 litre) (Figure 6a) and the sediment samples were collected by a small van Veen grab (0.025 m^2) (Figure 6b). The samples were handled with powder free nitrile gloves and without wearing perfume, deodorant, and body or suntan lotion.

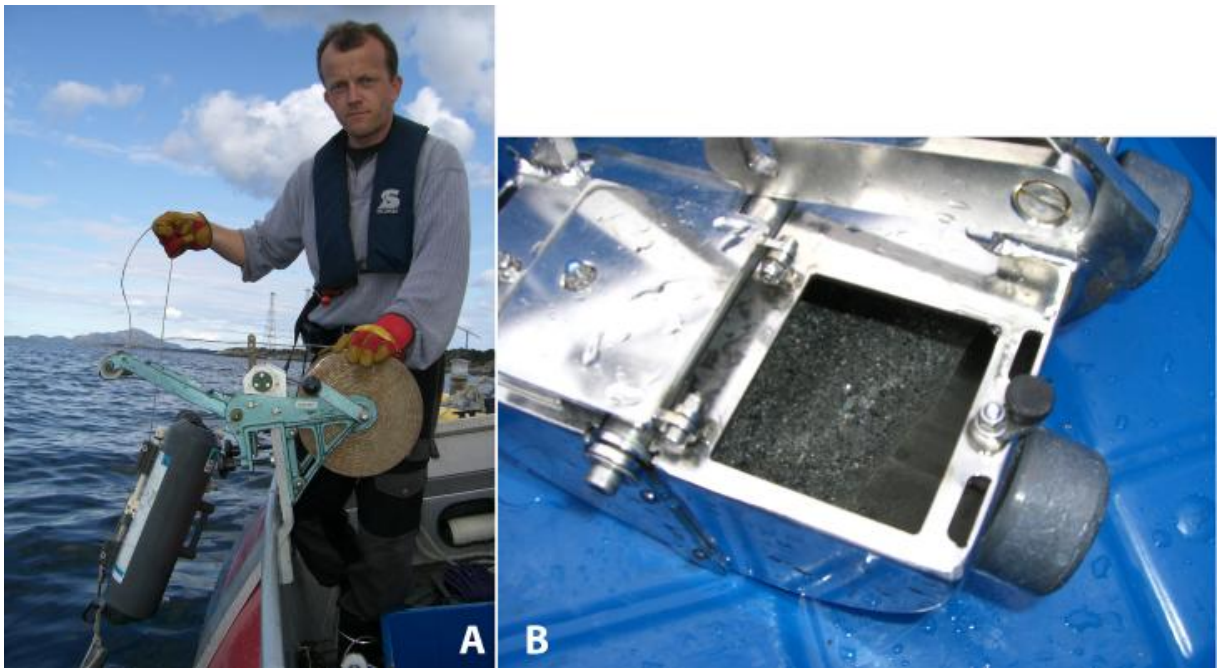

Figure 6: (a) Niskin water sampler, (b) Small van Veen grab.

4.5.1 Receiving water sampling (water, sediments, blue mussels)

The following samples were collected from the receiving waters at two wastewater treatment plant locations (WTP); VEAS in the inner Oslofjord and Breivika in the Tromsøsund:

- Sediments.
- Sea water outside diffuser/effluent site.
- Blue mussels close to the treatment site/location.

All the marine samples (sediments, water and blue mussels) were collected by NIVA except in Breivika in Tromsøsund where Akvaplan-niva did the fieldwork. In the inner Oslofjord, Bømlafjord and Breivika we used a small boat for fieldwork, except from the sediments outside VEAS were we used F/F Trygve Braarud (University of Oslo). In the Romsdalsfjord a boat owned by the fish farm was used.

4.5.2 Inner Oslofjord outside VEAS

The water was collected at the discharge depth of the effluent water. This depth is located between 25 to 30 m at VEAS [84], and the samples were taken at 28 m deep. The first station was located close to the diffuser/effluent point and, due to southward current in the area [84], the next sampling points were located at a distance of 100 m, 200 m, 300 m and 400 m south from the discharge point. One blank sample was collected at the first station. The water was sampled by the Niskin water sampler and it was immediately poured directly into the plastic and glass containers. Phosphoric acid (3 M) was added to the IVL-1 and IVL-2 samples until pH ~2. The water was stored dark and cooled (5 °C) before delivery to NILU at Kjeller who distributed the samples.

The sediments were collected at one station at a depth of 31 m (230 m from the centre of the diffusor at VEAS due to the security distance) and three replicates were taken. The surface sediment (0-2 cm) was collected and stored cool (5 °C) before delivery to NILU who distributed the samples.

At VEAS blue mussels (3 to 5 cm length) were collected at two stations on each side of the effluent; north at Gåsøya and south at Ramton (both also CEMP-stations). They were frozen immediately without being depurated. Each bulked sample contained 60 blue mussels and they were treated similarly as those for the Coordinated Environmental Monitoring Programme (CEMP - part of the Joint Assessment and Monitoring Programme JAMP), where OSPAR guidelines are used [85]. The samples were frozen when delivered to NILU and IVL. Details regarding the sampling of receiving water, sediment and blue mussel in Oslofjord are given in Table 7.

Table 7: Details regarding sampling of receiving water, sediment and blue mussel from Oslofjord.

Area	Station	Sample type	Sampling depth (m)	Sampling equipment	Sample date	Analysis
Inner Oslofjord	Slemmestad bank	Water + blank 1	25	Niskin water sampler	25.08.2008 28.08.2008	NIVA-1; NILU-1,2,3; IVL-1,2
Inner Oslofjord	Slemmestad bank	Water 2	25		25.08.2008	NIVA-1; NILU-1,2,3; IVL-1,2
Inner Oslofjord	Slemmestad bank	Water 3	28		25.08.2008	NIVA-1; NILU-1,2,3; IVL-1,2
Inner Oslofjord	Slemmestad bank	Water 4	28		25.08.2008	NIVA-1; NILU-1,2,3; IVL-1,2
Inner Oslofjord	Slemmestad bank	Water 5	28		25.08.2008	NIVA-1; NILU-1,2,3; IVL-1,2
Inner Oslofjord	Slemmestad bank	Sediment, grab 1	31	Small vanVeen grab	14.08.2008	NIVA-1; NILU-1,3; IVL-1
Inner Oslofjord	Slemmestad bank	Sediment, grab 2	31		14.08.2008	NIVA-1; NILU-1,3; IVL-1
Inner Oslofjord	Slemmestad bank	Sediment, grab 3	31		14.08.2008	NIVA-1; NILU-1,3; IVL-1
Inner Oslofjord	Gåsøya	Blue mussels 1	surface	Net cage	17.06.2008	NIVA-1; NILU-1; IVL-1
Inner Oslofjord	Ramton	Blue mussels 2	surface		17.06.2008	NIVA-1; NILU-1; IVL-1

4.5.3 Tromsøsund outside Breivika WTP

The coordinates of the effluent site was given by the municipality of Tromsø. The water was collected at a depth of 28 m right above the effluent site and then at distances of 50 m, 100 m 150 m and 250 m. The tidal current was strong (so the dilution may be high), and the samples were collected in the main direction of the current. One blank sample was collected at the second station (50 m from the source). Phosphoric acid (3 M) was added to the IVL-1 and IVL-2 samples until pH ~2. The water was stored dark and cooled (5 °C) before delivered to NILU Tromsø who distributed the samples.

The sediments were collected at one station at a depth of 30 m and three replicates were taken. The surface sediment (0 - 2 cm) was collected and delivered to NILU in Tromsø for distribution.
The blue mussels (3 to 6 cm length) were collected from two stations close to the effluent site; one north of the site and the other close to the site. The blue mussels were stored frozen without being depurated and sent to NIVA where they were made into bulked samples (60 mussels) before they were sent frozen to IVL and NILU.
Details regarding the sampling of receiving water, sediment and blue mussel in Oslofjord are given in Table 8.

Table 8: Details regarding sampling of receiving water, sediment and blue mussel from Tromsøsund.

Area	Station	Sample type	Sampling depth (m)	Sampling equipment	Sample date	Analysis
Tromsøsund	Tromsø strait	Water	28	Niskin water sampler	23.09.2008	NIVA-1; NILU-1,2,3; IVL-1,2
Tromsøsund	Tromsø strait	Water + blank	28		23.09.2008	NIVA-1; NILU-1,2,3; IVL-1,2
Tromsøsund	Tromsø strait	Water	28		25.09.2008	NIVA-1; NILU-1,2,3; IVL-1,2
Tromsøsund	Tromsø strait	Water	28		25.09.2008	NIVA-1; NILU-1,2,3; IVL-1,2
Tromsøsund	Tromsø strait	Water	28		25.09.2008	NIVA-1; NILU-1,2,3; IVL-1,2
Tromsøsund	Tromsø strait	Sediment	30	Small van Veen grab	10.09.2008	NIVA-1; NILU-1,3; IVL-1
Tromsøsund	Tromsø strait	Sediment	30		10.09.2008	NIVA-1; NILU-1,3; IVL-1
Tromsøsund	Tromsø strait	Sediment	30		10.09.2008	NIVA-1; NILU-1,3; IVL-1
Tromsøsund	Tromsø strait	Blue mussels	surface	Net cage	10.09.2008	NIVA-1; NILU-1; IVL-1
Tromsøsund	Tromsø strait	Blue mussels	surface		10.09.2008	NIVA-1; NILU-1; IVL-1

4.6 Fish farm sampling (water, sediments, blue mussels)

The following samples were collected:

- Sediment samples below the net cage of the fish farm.
- Sea water samples close to the net cages of the fish farm in increasing distance.
- Blue mussels close to the fish farm.

4.6.1 Fish farm 1 and 2

At the two fish farm sites 1 sediment station (five replicates) and five water samples with increasing distance relative to station 1, and two stations of blue mussels were collected.
The water was sampled at five stations with increasing distance from station 1 at 10 m depths. One blank sample was collected at station 1. The water was stored dark and cooled (5 °C) before delivery to NIVA for analysis.
Five replicate sediments were collected from station 1 by a small van Veen grab. The surface sediment (0-2 cm) was collected and stored cool (5 °C) before delivery to NIVA for analysis.

Blue mussels were collected the same way as in the receiving waters. At the Bømlafjord site (fish farm 1) blue mussels (3 to 5 cm length) were collected north of the fish farm on a fish net that was no longer in use, and south of the fish farm at a buoy close by. At the Romsdalsfjord site (fish farm 2) blue mussels (3 to 5 cm length) were collected 50 m and 100 - 150 m from the fish nets in eastern direction. All blue mussels were delivered frozen to NIVA for analysis. The details regarding sampling of surface water, sediment, and blue mussel from fish farms 1 and 2 and are given in Table 9 and Table 10, respectively.

Table 9: Details regarding sampling of surface water, sediment and blue mussel from fish farm 1.

Area	Station	Sample type	Sampling depth (m)	Sampling equipment	Sample date	Analysis
Bømlafjord	Fish farm 1	Water + blank 1	10		08.09.2008	NIVA-2
Bømlafjord	Fish farm 1	Water 2	10	Niskin	08.09.2008	NIVA-2
Bømlafjord	Fish farm 1	Water 3	10	water	08.09.2008	NIVA-2
Bømlafjord	Fish farm 1	Water 4	10	sampler	08.09.2008	NIVA-2
Bømlafjord	Fish farm 1	Water 5	10		08.09.2008	NIVA-2
Bømlafjord	Fish farm 1	Sediment, grab 1	44		08.09.2008	NIVA-2
Bømlafjord	Fish farm 1	Sediment, grab 2	44	Small	08.09.2008	NIVA-2
Bømlafjord	Fish farm 1	Sediment, grab 3	44	van Veen	08.09.2008	NIVA-2
Bømlafjord	Fish farm 1	Sediment, grab 4	44	grab	08.09.2008	NIVA-2
Bømlafjord	Fish farm 1	Sediment, grab 5	44		08.09.2008	NIVA-2
Bømlafjord	Fish farm 1	Blue mussels 1	surface	Net cage	08.09.2008	NIVA-2

Table 10: Details regarding sampling of surface water, sediment and blue mussel from fish farm 2.

Area	Station	Sample type	Sampling depth (m)	Sampling equipment	Sample date	Analysis
Romsdalsfjord	Fish farm 2	Water + blank 1	10		18.09.2008	NIVA-2
Romsdalsfjord	Fish farm 2	Water 2	10	Niskin	18.09.2008	NIVA-2
Romsdalsfjord	Fish farm 2	Water 3	10	water	18.09.2008	NIVA-2
Romsdalsfjord	Fish farm 2	Water 4	10	sampler	18.09.2008	NIVA-2
Romsdalsfjord	Fish farm 2	Water 5	10		18.09.2008	NIVA-2
Romsdalsfjord	Fish farm 2	Sediment, grab 1	30	Small	18.09.2008	NIVA-2
Romsdalsfjord	Fish farm 2	Sediment, grab 2	30	van Veen	18.09.2008	NIVA-2
Romsdalsfjord	Fish farm 2	Sediment, grab 3	30	grab	18.09.2008	NIVA-2
Romsdalsfjord	Fish farm 2	Sediment, grab 4	30		18.09.2008	NIVA-2
Romsdalsfjord	Fish farm 2	Sediment, grab 5	30		18.09.2008	NIVA-2
Romsdalsfjord	Fish farm 2	Blue mussels 1	surface	Net cage	18.09.2008	NIVA-2

4.7 Chemical analysis

4.7.1 Selected human pharmaceuticals (NIVA-1)

Chemicals
All HPLC solvents were purchased from Rathburn Chemicals Ltd (Scotland, UK). All pharmaceutical standards were of high purity (> 90%) and, with the exception of atorvastatin; they were purchased from Sigma-Aldrich (Steinheim, Germany). Atorvastatin was purchased from Mikromol GmbH (Germany).

Analytes
Propranolol, spiramycin, sertraline, paracetamol, atorvastatin, naproxen, amitriptyline, morphine, tamoxifen, warfarin and carbamazepine were simultaneously extracted and analysed.

Aqueous phase extraction
Effluent samples (approximately 2.5 L) were filtered (0.45 µm GFC) prior to extraction and seawater samples were untreated. 100 ng of internal standard (d_4-fluoxetine, d_2-phenacetin and ^{13}C-tamoxifen) was added to all samples before solid phase extraction (SPE). StrataX SPE columns (200 mg; Phenomenex) were conditioned by the addition of 5 mL methanol followed by 5 mL water. After conditioning, the sample was applied to the column under vacuum at a flow rate of approximately 2 mL/min. The column was air dried for approximately 30 minutes before analyte elution into silanised glass tubes. Elution used MeOH (6 mL), MeOH (2% acetic acid) (6 mL) and finally MeOH (2% ammonium hydroxide) (6 mL). Eluants were combined and then evapour ated under nitrogen to approximately 100 µL and reconstituted with methanol up to 1 mL. A blank and a spiked reference sample were extracted alongside each batch of samples.

Sludge, sediment and biota extraction
100 ng of internal standard (d_4-fluoxetine, d_2-phenacetin and ^{13}C-tamoxifen) was added to approx 1 g of freeze dried sludge/sediment sample and approx 5 g wet biota sample. Samples were mixed with hydro-matrix and extracted by accelerated solvent extraction using a modification of a previously reported method [28]. The modified method consisted of pre-fill method: methanol/water (1:1); equilibration, 5 min; static time, 5 min; flush volume, 60%; purge time, 60 s; static cycles, 3; and temperature 80 ^{o}C. Extracts were evapour ated to approx 5 mL under nitrogen and reconstituted with ultrapure water to 1 L into silanised glass bottles. Extracts were then cleaned up using the aqueous phase extraction method above.

LC-MS/MS analysis
Liquid chromatography – mass spectrometry (LC-MS) analysis used a Waters Aquity UPLC coupled to a Waters Quattro Premier XE triple quadruple mass spectrometer. Analytes were separated on an Aquity BEH C18 1.7 µm column (2.1 × 50 mm) (Waters, Sweden). The mobile phases for optimised separation were modified water (10 mM ammonium acetate) and modified methanol (10 mM ammonium acetate). Gradient elution gave good separation of all compounds at a flow rate of 0.35 mL/min.

Standards (100 µg/mL) were made in methanol and directly infused into the MS to optimise MS parameters. Warfarin was detected in negative ion mode and all other analytes were detected in positive mode. The capillary was set to 3 kV, the source temperature 100 ^{o}C and the desolvation temperature 350 ^{o}C. The nitrogen cone gas was at a flow rate of 50 L/hr and

the argon desolvation gas at 1000 L/hr. After separation and MS optimisation, extracted matrix was spiked with standards to investigate any matrix interferences.

4.7.2 Selected hospital-use pharmaceuticals 1; Antibiotics (NILU-1)

Determination of ß-lactam antibiotics
The ß-lactam antibiotics amoxicillin, cefotaxime, cefalotin, meropenem, ofloxacin, penicillin-G and pivmecillinam were analyzed by Ultra Pressure Liquid Chromatography –Time-Of-Flight high resolution mass spectrometry (UPLC-TOF-HRMS).

Sample preparation
Water samples: Aliquots of sea water (1000 mL) or sewage water (400 mL) were adjusted to pH 8 and added isotope labelled amoxicillin and penicillin G as internal standards. The samples were extracted by mixed-mode solid phase extraction (SPE) using a polymer/anion exchange sorbent. After elution with an acidified solvent, the samples were further cleaned by dispersive SPE using C18 material and finally concentrated using nitrogen.

Sediment, sludge, mussels: The samples were extracted by shaking with acetonitrile/water. After centrifugation an aliquot of the extract was cleaned by dispersive SPE using C18 material. Finally, the samples were concentrated under nitrogen before UPLC-TOF analysis.

Instrumental analysis
The antibiotics were separated on a Waters Acquity UPLC equipped with a reversed phase phenyl column, Waters UPLC BEH Phenyl, 100 × 2.1 mm i.d., 1.7 µm particle size. Acetonitrile and purified water acidified with formic acid was used as the mobile phase. The compounds were detected on Waters LCT Premier TOF-MS using electro spray ionization in positive, high resolution mode. Quantification was performed using isotope labelled internal standards.

4.7.3 Selected hospital-use pharmaceuticals 2; X-ray contrast agents (NILU-2)

Extraction of aqueous samples
Aqueous phase samples were stored on amber glass bottles and the water samples (200-400 mL) were extracted onto StrataX cartridges (Phenomenex). Elution was performed by 10 mL of a mixture of acetone/methanol, and the extract volume was reduced to 1 mL before analysis.

Extraction of sediments
2-4 gram of the sediment sample was extracted in 10 ml MilliQ water by sonication. The procedure was repeated three times. The sample extracts were combined and further treated as water samples.

LC-HRMS analysis
Liquid chromatography was performed with an Agilent 1100 liquid chromatography system (Agilent Technologies, Waldbronn, Germany), equipped with an auto sampler, a quaternary pump, an on-line degassing system. The compound separation was performed with a reversed phase C_{18} column (Atlantis dC18, 2.1 mm ID × 150 mm length, 3 µm, Waters, Milford USA). A stainless steel inlet filter (Supelco, 0.8 µm) was used in front of a pre-column with the same stationary phase as the separation columns. Gradient elution was performed with water as solvent A and acetonitrile as solvent B. The binary gradient had a flow rate of 0.2 mL/min and started with 100 % A. Solvent B was introduced linear up to 100% at 10 minutes with a

linear increase in flow rate up to 0.5 mL/min at 10 minutes. This setting was kept isocratic until 15 minutes with a subsequent equilibration of the column. The analytical detector was a Micromass LCT orthogonal-acceleration time-of-flight (TOF) mass spectrometer (MS) equipped with a Z-spray electro spray ion source and a 4 GHz time to digital converter (TDC) (Micromass Ltd., Wythenshawe, Manchester, UK). The instrument was operated in positive mode and the electro spray source parameters were optimised to the following values: sample cone 35 V, capillary voltage 3.1 kV, extraction cone 3 V, source temperature 125 °C, desolvation temperature 350 °C, cone gas flow 24 L/h and desolvation gas flow 700 L/h. The pusher frequency was operated in automatic mode. The data processing and instrument (LC-HRMS) control were performed by the MassLynx software. The quantitation was performed with signal extraction of a peak width of 100 mDa and the standard addition method. Details on the mass spectrometric detection on the x-ray contrast agents is given in Table 11

Table 11: Molecular ion, adduct ions and confirming ions.

Compound	MW	$[M + H]^+$	Confirming ion
Iodixanol	1549.7	1550.7	1571.7
Iohexol	820.9	821.9	843.7
Iopromide	790.9	791.9	813.7

4.7.4 Selected hospital-use pharmaceuticals 3; Cytostatics (NILU-3)

Chemicals

All solvents for HPLC and sample preparation were purchased from VWR (Darmstadt, DE). The deuterated internal standards d_{10}-irinotecan and d_9-docetaxel and the metabolites doxorubicinol and 6-OH-paclitaxel were purchased from Toronto Research Chemical (Toronto, Canada). The pharmaceuticals bortezomib, docetaxel, doxorubicin, irinotecan, and paclitaxel were purchased through the hospital pharmacy at UNN. Due to governmental requirements for proper handling of cytostatics, the dilution of all compounds was conducted by qualified pharmacists at UNN.

Aqueous phase extraction

Hospital and STP effluent water (1 L) and receiving water (2.5 L) samples were filtered (0.45 µm) prior to extraction. 100 µg of each internal standard was added to all samples and the samples were shaken (120 min^{1}) for 30 min prior to SPE. Oasis HLB (Waters) SPE columns (300 mg) were conditioned by 7 mL acetone, 7 mL methanol, and 5 mL MilliQ water, and the samples were then applied to the column at a flow of 0.5 - 2 mL/min. The columns were washed with 5 mL 0.25% aqueous NH_4OH also containing 5% methanol (v/v) and 5 mL hexane, and were then air dried for 30 min. The analytes were eluted with 7 mL methanol (1% HCOOH) and 5 mL acetone (1% HCOOH). The solvents were removed under reduced pressure to a residual volume of ~200 µL and approximately 200 µL methanol and 500 µL MilliQ water was added.

Sludge and sediment extraction

Sludge and sediment samples were freeze-dried for 48 h before extraction. 200 µg of each internal standard was added to 0.5 g sample, and the samples were rested for 30 min. The analytes were extracted with 3 × 7mL MeOH:HCOOH (99:1, v/v) under sonication for 30 min. The samples were centrifuged (2500 min^{1}) and the methanol decanted off for each step. The combined methanol phases were removed under reduced pressure to ~1 mL. 1 mL MilliQ water was added and the samples were filtered (0.22 µm) and transferred to a HPLC injector vial.

LC-MS analysis
Liquid chromatography-mass spectrometry was used for the analysis of cytostatics. The instruments were a 1525µ pump, a 2777 auto sampler and a QTOF micro, all from Waters (Bedford, MA). The analytes were separated on a Synergi MaxRP C18 column (75 × 2.1 mm; 4 µm) from Phenomenex (Torrance, CA). Gradient elution gave satisfactory separation of all compounds at a flow rate of 0.2 mL/min. 20 µL was injected.

Individual standard solutions of the analytes in methanol were infused into the MS to optimize MS variables. All analytes were detected in positive electro spray mode at +3 kV. The source and desolvation temperature was 120 and 400 °C, respectively. Nitrogen was used as cone (50 L/hr), desolvation (600 L/hr), and nebulizing (max flow) gas. The limits of detection for the cytostatics are given in appendix 3, and the analyte recovery was 44 - 76%.

4.7.5 Selected aquaculture medicines (NIVA-2)

Analytes
Emamectin, praziquantel, oxolinic acid, fenbendazole and flumequine were simultaneously extracted and analysed in sediment and biota. The pyrethroids, cypermethrin and deltamethrin were extracted and analysed together in sediment and biota. All analytes were extracted and analysed simultaneously in aqueous phase samples.
Aqueous phase extraction for all analytes
Receiving water samples (approximately 2.5 L) were extracted by solid phase extraction using 200 mg OASIS HLB columns (Waters, Sweden). The columns were conditioned by the addition of 5 mL methanol followed by 5 mL water. After conditioning, the samples were applied to the column under vacuum at a flow rate of approximately 2 mL/min. The column was air dried for approximately 30 minutes before analyte elution into silanised glass tubes. Elution used MeOH (6 mL), MeOH (2% acetic acid) (6 mL) and finally MeOH (2% ammonium hydroxide) (6 mL). Eluants were combined and then evapour ated under nitrogen to approximately 100 µL and reconstituted with methanol up to 1 mL. A blank and a spiked reference sample were extracted alongside each batch of samples. 100 µL was removed and solvent exchanged to cyclohexane in preparation for pyrethroid analysis by GC/ECD.

Sediment and biota extraction
Pyrethroids. Approx 5 g of freeze dried sediment and 7 g of wet biota samples were double solvent extracted with DCM. 10 mL DCM was added to each samples and sonicated at 60 ^{o}C for 30 min. Samples were centrifuged at 3000 rpm for 10 min and the DCM eluant decanted. This was repeated and the eluants combined before evapour ation under nitrogen and solvent exchange to approximately 1 mL cyclohexane. Potential interferences were removed by treating the extract with 1 mL concentrated sulphuric acid. Centrifuging at 2000 rpm for 5 min ensured complete separation of the acid and solvent layer.
Quinolones and anthelmintics. Approximately 2 g of freeze dried sediment and 5 g of wet biota were extracted by accelerated solvent extraction. The method consisted of pre-fill method: acetonitrile/water (7:3) (0.2% formic acid); equilibration, 5 min; static time, 5 min; flush volume, 60%; purge time, 60 s; static cycles, 3; and temperature 100 ^{o}C. Extracts were evapour ated to approx 5 mL under nitrogen and reconstituted with ultrapure water to 1 L into silanised glass bottles. Extracts were then cleaned up using the aqueous phase extraction method above.

GC-ECD analysis of pyrethroids
Cypermethrin and deltamethrin analysis was performed on a Hewlett-Packard 6890 GC fitted with a ^{63}Ni µECD detector. The injector was operated in splitless mode (1.25 mins) at 255 ^{o}C.

The separation of the pyrethroids was performed on a DB-5 column (60 m × 0.25 mm, 0.25 µm film thickness (J&W Scientific). The GC oven temperature was programmed as follows: 90 ^{o}C held for 2 mins, 20 ^{o}C/min to 180 ^{o}C, 2 ^{o}C/min to 270 ^{o}C, 20 ^{o}C/min to 310 ^{o}C and held for 5 mins. Hydrogen was the carrier gas at a flow rate of 1 mL/min and nitrogen was used as the make-up gas at a flow rate of 30 mL/min with the detector temperature set to 285 ^{o}C. For quantification, the peak areas of the 4 cypermethrin isomers were added together.

LC-MS/MS analysis of quinolones and anthelmintics

For LC-MS analysis the same equipment as for the analysis of pharmaceuticals was used. Analytes were separated on an Aquity BEH C18 1.7 µm column (2.1 × 50 mm) (Waters, Sweden). The mobile phases for optimised separation were modified water (0.1% formic acid) and modified methanol (0.1% formic acid). Gradient elution gave good separation of all compounds at a flow rate of 0.3 mL/min.

Standards (100 µg/mL) were made in methanol and directly infused into the MS to optimise MS parameters. All analytes were detected in positive mode. The capillary was set to 3 kV, the source temperature 120 ^{o}C and the desolvation temperature 350 ^{o}C. The nitrogen cone gas was at a flow rate of 50 L/hr and the argon desolvation gas at 700 L/hr. After separation and MS optimisation, extracted matrix was spiked with standards to investigate any matrix interferences. The limits of detection for the analytes are given in appendix 3, and the analyte recovery was 60-153%.

4.7.6 Determination of EDTA (IVL-4)

Water samples

Water samples (50 mL) were analysed with regard to EDTA after filtration (pre-heated GF/C-filter). The sample was spiked with surrogate standards and subsequently concentrated on an SPE-column (Isolute ENV+; ~15 mL/min). After the sample had passed through, the column was rinsed with diluted HCl and subsequently dried for approximately 15 min under vacuum. The analytes were eluted and the eluate was evapour ated to dryness.

The acids in the eluate were esterified to the corresponding propylesters by the reagent propanol/HCl at 90 °C for 1 hour. The reaction was terminated by adding a carbonate buffer and the derivatives were extracted with hexane. The hexane phase was withdrawn, dried over sodium sulphate and concentrated under nitrogen gas. Prior to gas chromatography using a Nitrogen Phosphorus Detector (GC-NPD), a volumetric standard was added.

Sediment and sludge samples

Freeze-dried sediment or sludge samples (~0.5 g) were spiked with recovery standards and mixed well. After addition of zinc sulphate and ultra pure water the sample was treated in an ultra sonic bath for 15 min. Phosphate solution (KH_2PO_4) was added and the sample was again treated in the ultrasonic bath (5 min) and then gentle agitated on a shaking board (30 min). After centrifugation the clear water was extract safeguarded. The extraction cycle was repeated twice with ultra pure water and the extracts were combined. After acidification the water sample was concentrated and cleaned up on two different solid phase columns in series. The eluate from the columns was thereafter treated in the same way as the eluate from the water samples.

Sea mussel samples

The soft tissue (1 g f.w.) of the mussels were thawed and dried at 105 °C overnight. The dry weight was determined and the sample grinded using mortar and pestle. Derivatisation reagent was added and the reaction was performed at 90 °C for 1 hour. The reaction was

terminated by adding a carbonate buffer and the derivatives were extracted with hexane. The hexane extract was subjected to a cleanup protocol implying liquid-liquid extraction in order to eliminate interfering matrix substances. The hexane phase was withdrawn, dried over sodium sulphate and concentrated under nitrogen gas. Prior to GC-NPD analysis a volumetric standard was added.

Instrumentation
The analysis was carried out with a HP 5890 Series II GC-NPD system, on-column injector and a HP 7376 auto sampler, all from Hewlett-Packard. The column consisted of two parts: (a) a of methyl deactivated megabore pre-column (0.53 µm, 10-15 cm) needed for the auto on-column injector, (b) an analytical fused silica capillary column (15 m) with an ID of 0.25 mm and a film thickness of 0.25 µm (RTX-5 MS; Restek;). After 50-100 injections, or when peak tailing appeared, the megabore part was exchanged. The following temperature program was used: 1 min isothermal at 100°C followed by an increase of 25°C/min to 200°C and then 10°C/min to 300°C, hold for 20 min. The detector signal from the gas chromatograph was acquired and processed with the chromatography data program TurbochromTM. The compounds were identified and quantified by comparison of their retention time and peak area to authentic reference compounds. The recovery of the analyte was estimated by means of the added surrogate standards.

4.7.7 Determination of diethyl phthalate (DEP), butyl paraben and avobenzone (IVL-3)

The analysis of these compounds was divided in two parts: (a) determination of DEP and butyl paraben after acetylation according to Remberger [86] and (b) determination of avobenzone after subsequent methylation with sodium hydride/methyl iodide according to Nagtegaal [87].

Water samples
The water samples (200-800 mL) were filtrated (pre-heated GF/C-filter) prior to solid phase extraction. The sample was then spiked with surrogate standards and subsequently acidified and concentrated on the SPE-column (~15 mL/min). After the sample had passed through, the column was rinsed with water and subsequently dried. The analytes were eluted with methanol and a mixture of hexane:MTBE. The extracts were combined and the methanol was washed away by shaking the extract with water. The extract was dried over sodium sulphate and acetylated with the reagent acetic acid anhydride using sodium acetate as base. The reaction was terminated by adding a carbonate buffer and the derivatives were withdrawn and used for the determination of butyl paraben and diethyl phthalate.

After the determination of these compounds the solvent was exchanged to molecular sieve (4 Å) dried MTBE. The reagent sodium hydride and methyl iodide were added and the methylation was performed for 2 hours at 85°C. After chilling water was carefully added to the reaction mixture (violent exothermic reaction) followed by hexane. The sample was extracted and the extract was used for the determination of avobenzone.

Sediment and sludge samples
Sediment (10 g f.w) or sludge (2 g f.w) was acidified with phosphorus acid and extracted twice with acetone:hexane (1:1) first in an ultra sonic bath (5 min) and then on a shaking board (25 min). The acetone was removed from the combined organic extract by shaking it with acidified water. The extract was acetylated (see water samples) and applied onto a silica gel column. Two fractions were collected: (a) hexane and (b) hexane:MTBE (9:1). The former fraction was discarded and latter was used for the determination of butyl paraben and diethyl

phthalate. The fraction used for the determination of butyl paraben and diethyl phthalate was also used for the determination of avobenzone after methylation (see water analysis).

Sea mussel samples

Homogenised mussel sample (5 g f.w.), fortified with recovery standard was extracted with hexane:acetone in an ultra sonic bath (5 min) and on a shaking board for 25 min. The solvent extract was, after centrifugation, transferred to a separator funnel and the acetone was removed by shaking the extract with KH_2PO_4-buffer. The extract was evapour ated to dryness using nitrogen gas and the lipid content was determined by weighing. Hereafter the extract was treated in the same way as for sludge samples (see above).

Instrumentation

The sample extracts were analysed on a 6890N gas chromatograph coupled to a 5973N mass selective detector (Agilent). The injection, 1 µL, was done in splitless mode at 240°C. The fused silica capillary column (VF-5MS 30 m × 0.25 mm i.d. × 0.25 µm film thickness, Varian) was held at 45°C for 1 min., ramped 15°C/min to 200°C, 5°C/min until 300°C and held at 300°C for 5 min. Helium was used as carrier gas. The detector was used in selected ion monitoring mode (SIM) with electron ionisation energy of 70 eV. The analytes were identified by their characteristic retention time and one quantification ion (Trg-ion) and one or two supporting ions (Q1-Q2-ion) used to increase specificity was recorded (Table 12).
Quantification was based on comparison of peak abundance to the known response of an internal standard. The reported analyte concentrations were corrected according to the determined surrogate standard losses.

Table 12: Ions used in MS analysis. Abbreviations: Names in italics are the recovery standards and injection standard (biphenyl). t_R: retention time; Trg: target ion; Q1 and Q2: qualifier ions.

Substance	t_R (min)	Trg-ion	Q1-ion	Q2-ion
Biphenyl (*Injection standard)*	9.75	154	-	-
Diethylphthalate	11.38	149	177	176
Butyl paraben e acetate	12.69	138	121	194
Avobenzone metylated	17.51	135	161	338
3-F-propylparaben *(Recovery standard)*	11.48	156	139	-
Dialylphthalate *(Recovery standard*)	10.96	111	169	-

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 min., (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 [88].
Field blanks were collected at each sampling station. A method blank was included for each sample batch analysed to assess background interferences and possible contamination of the samples. Concentrations below field blank levels are treated as not detected.
Possible background levels of analytes were subtracted from measured sample values [89, 90]

4.7.8 Analysis of Sodium dodecyl sulphate (SDS), Sodium laureth sulphate (SDSEO) and Cocoamidopropyl betaine (CAPB) (IVL-2)

Internal standard (4-Octylbenzene sulfonic acid, n-C8-LAS, Aldrich) was added to all samples. Water was, without previous filtration, extracted on a graphitized carbon black SPE column (Supelclean ENVI-Carb, Supelco), washed with methanol and eluted with

dichloromethane/methanol containing tetramethylammoniumhydroxide [91]. After evapour ation the extract was redisolved in equal parts 10 mM NH_4OAc in water and methanol and analyzed by LC-MS-MS.

Sediment
Freeze dried sediment was extracted with methanol. After centrifugation the extract was treated on a graphitized carbon black SPE column the same way as described for water samples and analyzed by LC-MS-MS.

Sludge
Freeze dried sludge was extracted with methanol. After centrifugation the extract was diluted with equal parts 10 mM NH_4Ac in water and methanol and analyzed by LC-MS-MS.

LC-MS/MS
Liquid chromatography was performed using a Prominence UFLC system (Shimadzu) with two pumps LC-20AD, degasser DGU-20A5, autosampler SIL-20ACHT and column oven CTO-20AC. A column (Ascentis C8 50 × 2.1 mm, particle size 5 µm, Supelco) was installed in the eluent flow line immediately upstream the autosampler. This made analyte peaks originating from the solvent/solvent system elute later than peaks from the sample. The analytical column was a Thermo HyPurity C8 50 mm × 3 mm, particle size 5 µm (Dalco Chromtech). The solvent was 10 mM NH_4OAc in water mixed with methanol in a linear gradient from 30% to 100%. The column temperature was 50 °C and the flow rate 0.5 mL/min. The effluent was directed to an API 4000 triple quadrupole mass spectrometer (Applied Biosystems). Electrospray ionisation in negative mode was used. Precursor ion was the deprotonated molecular ion. Product ions were *m/z* 170 for [n-C8-LAS] and *m/z* 97 [SO_4H] and 80 [SO_3] for SDS.

Sodium dodecyl sulphate (SDS) was obtained from Sigma. Sodium laureth sulphate contains SDS and ethoxylated analogues. As individual ethoxylated compounds were not available a technical product (Chemos GmbH) was used. The sensitivity for the MRM transition molecular ion to *m/z* 97 [HSO_4] was assumed to be the same for the different ethoxylate chain lengths. By this assumption the following composition was found for the technical blend: SDS 21%, $SDSEO_1$ 27%, $SDSEO_2$ 31%, $SDSEO_3$ 15%, $SDSEO_4$ 6%. Sodium laureth sulphate concentration was calculated as the sum of SDSEO1, SDSEO2, SDSEO3 and SDSEO4. Cocoamidopropyl betaine was obtained as a 30% solution (Chemos). Precursor ion was *m/z* 341 [C12-CAPB H] and the product ion was *m/z* 102 [$(CH_3)_2NCH_2COO$] [92].

4.7.9 Analysis of Cetrimonium salt (IVL-1)

Water (25 mL) was acidified and 50 µg C12LAS was added. The sample was extracted with chloroform which was evapour ated to dryness [93]. The residue was re-dissolved in methanol and analyzed by LC-MS-MS.

Freeze dried sediment or sludge was extracted with concentrated hydrochloric acid diluted with methanol to a concentration of 1M in an ultrasonic bath (3 min) and then at 85°C (10 min). The extraction was repeated twice, the extract combined and the volume reduced to a few millilitres. After washing with hexane+MTBE (1:1) the extract was further evapour ated to dryness [57, 86]. The residue was dissolved in water (5 mL) containing 50 µg C12LAS. The solution was extracted with chloroform which was evapour ated to dryness, the residue re-dissolved in methanol and analyzed by LC-MS-MS.

Liquid chromatography-triple quadrupole mass spectrometry was performed as described above, but electrospray ionisation in positive mode was used. Trimethylhexadecylammonium chloride (ATAC-C16) (Sigma) was used as standard. Precursor ion was *m/z* 284, and product ion was *m/z* 60 $[(CH_3)_3NH]^+$.

4.8 Uncertainties

When performing environmental screening or monitoring all steps in the study starting with the design of the study, selection of sampling sites and sampling frequency, time of sampling, performing of sampling, transport and storage of samples, chemical analysis and data treatment are generating some degree of uncertainty. To quantitatively estimate the contribution of all steps is an extreme difficult task or not possible at all. However, we will discuss the relevance of the different contributors in a qualitative way.

One important question is whether a sample is representative for a given time period or a given region. Many of the selected compounds are semi-continuously emitted to the environment and a constant concentration of these compounds in the environment is not expected. Seasonal variations in the use of avobenzone (a UV-protecting agent with presumably fewer people sunbathing in the sample period than in the warmer days in the summer of 2008) will have severe influence on the measured environmental concentrations. The cytostatics (and probably also some anti-biotics) are only used in given periods, and it is not known whether the cytostatics covered in this screening actually were used in the sample period. In this screening, the samples were collected within a narrow time frame at (for each sample type) and at only two different geographical locations. The results obtained here are therefore only a snapshot of the reality at those two places at the given time.

Factors with influence on sampling uncertainty are analyte loss due to adsorption to sample containers, waste water flow and particle content, tidal water current, contamination (for some compounds), selection of sample type (water with or without particle phase), and degradation during transport and storage.

The uncertainty of the chemical analysis is governed by loss during extraction and clean-up, interference from other compounds, trueness of analytical standards, instrumental parameters, and contamination. A normal approach to estimate and quantify these factors is the participation in a laboratory intercalibration. However, at this stage the analysis of these compounds in environmental samples is not done routinely and intercalibration studies have not been available. The uncertainty is expected to be larger for compounds which are analysed the first time than for compounds which previously have been analysed or where similar compounds have been analysed earlier. That means that compounds like EDTA, paracetamol, or butyl paraben will probably have analytical uncertainty in the range of 20 to 40 %, whereas compounds like the cytostatics or detergents will probably have a higher analytical uncertainty 30 to 50 %. For all analytes we consider the analytical uncertainty as fit-for-purpose (that means adequate for a first screening study), however, the results cannot be implemented uncritically in time-trend studies.

5. Results and discussion

5.1 Pharmaceuticals and personal care products as environmental contaminants

In this chapter the results from this screening are presented along with any previously reported environmental presence of the individual compounds, any known ecotoxicological effects and information on environmental fate. The ecotoxicological effects known today do not follow the division of the compounds as personal care products, aquaculture medicines or pharmaceuticals. It is therefore only reasonable to discuss the environmental impact of the investigated PPCPs individually. For each compound a concluding remark regarding their environmental concern is provided along with a comment on their detected concentrations in this screening compared to previously reported concentrations. The complete results for all samples are presented in Appendix 3.

At the measured environmental concentrations acute toxic effects of the investigated PPCPs to aquatic organisms are unlikely to occur. However, many aquatic species are continuously exposed over long periods of time or even over their entire life cycle. Evaluation of the chronic potential of PPCPs is therefore important. Unfortunately, there is a lack of chronic data [8]. The available chronic data often do not cover the important key targets. Furthermore, toxicity experiments are usually performed according to standardized guidelines only. More specific investigations including analysis of possible targets of the PPCP are lacking, or have only rarely been performed. Life-cycle analyses are not reported and toxicity to benthic organisms has rarely been evaluated [8].

Information regarding the ecotoxicological effects of mixtures of compounds is even more scarce than for chronic effects [121]. Because current environmental risk assessments focus on single substances only, it is very likely that the prevailing assessments underestimate the real environmental impacts [10]. Additive effects may be expected in non-target organisms. Even synergistic effects have been reported for nonsteroid anti-inflammatory pharmaceutical exposure to *Daphnia* [122].

In the present study, only two metabolites were included. Most pharmaceuticals are transformed to more polar metabolites *in vivo*, and the Ecotoxicological effects of metabolites are for most compounds, unknown. Furthermore, genetic diversity within a species may render some individuals very sensitive to certain xenobiotics, but the knowledge on this topic is almost non-existent.

A standard approach for Ecotoxicological effects classification is the EEC criteria (Directive 93/67/EEC) which classify compounds according to their hazard to aquatic organisms (see Table 13 [46]).

Table 13: The EEC (Directive 93/67/EEC) Ecotoxicological effects classification.

Ecotoxicological effects	**EC_{50} (ng/L)**
'Very toxic'	<1 000 000
'Toxic'	1 000 000 - 10 000 000
'Harmful'	10 000 000 - 100 000 000

According to this, cefotaxime and meropenem are nontoxic, cypermethrin, deltamethrin, doxorubicin, EDTA, iodixanol, iohexol, and iopromide are harmful, and butyl paraben and emamectin are toxic to aquatic organisms. The remaining compounds are all defined as very toxic to aquatic organisms. A major shortcoming with this approach is that it does not take the observed concentrations into consideration, and it will thus not be further used.

In the discussion on the environmental concerns with the identified pharmaceuticals in the present study, the following criteria have been applied:

(i) If the compound was not detected or only detected in waste water, the compound was assessed to be of no or little environmental concern.

(ii) For compounds detected in receiving water and/or sediment, its highest detected concentration was compared with the worst case Ecotoxicological effects concentration found in the scientific literature:
   a. If the difference between highest observed concentration and the worst case Ecotoxicological effects concentration found in the scientific literature was more than 100 000, the compound was assessed to be of little or no environmental concern.
   b. If the difference between highest observed concentration and the worst case Ecotoxicological effects concentration found in the scientific literature was more than 1 000, but less than 100 000, the compound was assessed to be of *some environmental concern.*
   c. If the difference between highest observed concentration and the worst case Ecotoxicological effects concentration found in the scientific literature was less than 1000, the compound was assessed to be of *environmental concern*. 1000 was chosen as a safety factor as this often is applied as a safety factor in environmental risk assessments

(iii) Compounds identified in biota are automatically of environmental concern.

5.2 Selected human pharmaceuticals

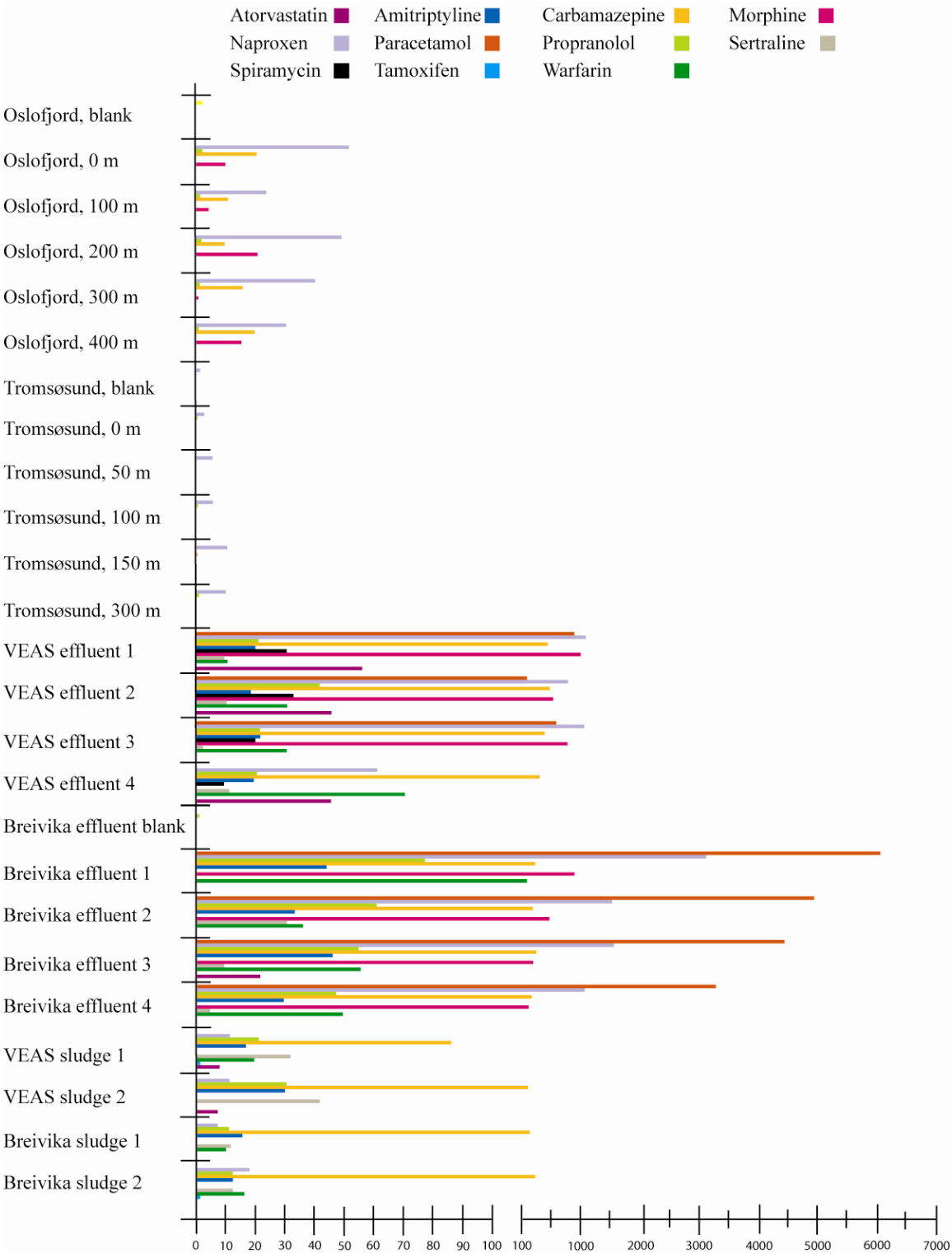

Figure 7: The figure shows the measured concentrations of Amitriptyline (trace ■), atorvastatin, (trace ■), carbamazepine (trace ■), morphine (trace ■), naproxen (trace ■), paracetamol (trace ■), propranolol (trace ■), sertraline (trace ■), spiramycin (trace ■), tamoxifen (trace ■), and warfarin (trace ■). The concentrations are presented as ng/L for aqueous samples and as ng/g (d.w.) for solid samples.

Amitriptyline

Results from this study

The detected amounts of amitriptyline are presented graphically in Figure 7, trace ■. Amitriptyline was detected in only one out of five receiving water in Oslofjord (1.1 ng/L), and in no samples from Tromsøsund (<LoD 1 ng/L), and amitriptyline was not detected in sediment samples (<LoD 1 ng/g d.w.) or mussels (<LoD 5 ng/g) from Tromsøsund and Oslofjord. Amitriptyline was not analysed in the Ullevål and UNN effluent samples.
Amitriptyline was detected in all STP effluent water samples in both Tromsø Breivika (30 - 45 ng/L) and VEAS (20 - 25 ng/L). Amitriptyline was also detected in sludge from VEAS (17 - 29 ng/g d.w.) and Breivika (13 - 16 ng/g d.w.).

Results from other studies

Amitriptyline has been detected at 3 ng/L in river water [5]. Amitriptyline has been detected at 849 ng/L [5] in STP influent water, and at 12.9 ng/L [6] and 207 ng/L [5] in STP effluent water.

Ecotoxicological effects

Some known ecotoxicological effects of amitriptyline are presented in Table 14

Table 14: Ecotoxicological effects of amitriptyline.

Species	End point/effect	Concentration (ng/L)	Reference
Mysidopsis bahia water flea	Chronic Toxicity Test EC_{50}	3 200 000	[3]
Cyprinodon variegates sheep head minnow	Chronic Toxicity Test EC_{50}	310 000	[3]
Pimephales promelas fathead minnow	Chronic Toxicity Test EC_{50}	320 000	[3]
Ceriodaphnia dubia (water flea)	Chronic Toxicity Test EC_{50}	1 000 000	[3]
Brachionus calyciflorus	Chronic Toxicity Test EC_{50}	81 000	[3]

An estimated BCF of 1,226 was calculated for amitriptyline, based on a log Kow of 4.92, suggesting the potential for bioconcentration in aquatic organisms is very high [37]. Amitriptyline has annual consumption rate of nearly 0.3 tonnes in Norway [3].

Fate

The compound degrades slowly in aqueous environments and have the potential to bioaccumulate [7].

Concluding remark

The levels of amitriptyline detected in this screening are comparable with previously reported levels. The detected concentration of 1.1 ng/L in receiving water is more than five orders of magnitude less than the lowest reported effect concentration for amitriptyline. Thus, the detected concentrations of amitriptyline do not cause environmental concern.

Atorvastatin

Results from this study

The detected amounts of atorvastatin are presented graphically in Figure 7, trace ■. Atorvastatin was not detected in any receiving water samples from Oslofjord or Tromsøsund (<LoD 1 - 2 ng/L), and atorvastatin was not detected in sediment samples (<LoD 5 ng/g d.w.) or in mussels from Tromsøsund and Oslofjord (<LoD 5 ng/g). Atorvastatin was not analysed in the Ullevål and UNN effluent samples. Atorvastatin was detected in three out of four STP effluent water samples from VEAS (45 - 56 ng/L) and in one out of four samples from

Tromsø Breivika (23 ng/L). Atorvastatin was detected in sludge from VEAS (8 - 10 ng/g d.w.), but not from Breivika (<LoD 5 ng/g d.w.).

Results from other studies
Atorvastatin has been detected in STP effluent water at 50-60 ng/L [8] and 22.4 ± 1.4 ng/L [9].

Ecotoxicological effects
The PBT Profiler software (www.pbtprofiler.net/) has estimated the chronic toxicity threshold of atorvastatin toward "fish" to be 86 000 ng/L (no observed effect concentration; NOEC) [94]. Statins are known inhibitors of sterol biosynthesis in plants and have displayed phytotoxicity in radish and aquatic plants of *Lemna* genus [10]. A study with *Lemna gibba* indicated that statins caused concentration-dependent toxicity via reduction of mevalonate (HMG-CoA mediated) derived products [10]. The acute toxicity of atorvastatin toward the midge *Chironomus tentans* and the fresh water shrimp *Hyalella azteca* using standard 10-d acute toxicity tests showed that atorvastatin was approximately 10 times more toxic to *Hyalella azteca* compared to *Ch. tentans* [10]. The measured toxicity thresholds were several orders of magnitude higher than current environmental concentrations, indicating that this compound poses little risk to invertebrates [10]. Some known ecotoxicological concentrations of atorvastatin are presented in Table 15.

Table 15: Ecotoxicological effects of atorvastatin.

Species	End point/effect	Concentration (ng/L)	Reference
Daphnia magna	EC_{50}, 48 h	200 000 000	fass.se
Water flea	NOEC, 48 h	81 000 000	fass.se
Lemna gibba	Decreased (50%) stigmasterol	36 000	[10]
Duckweed, plant	and sitosterol concentrations		
	EC_{10}	85 000	[18]
	EC_{10}	130 000	[95]

Fate
No atorvastatin remained after 6 h of UV exposure yielding a half-life of 3.5 hours. Therefore photo degradation is believed to be important for atorvastatin in aquatic environments [10]. The second order rate constant for photo degradation of atorvastatin is $1.9\pm0.5\times10^{10}$ M^{-1} s^{-1} [10].

Concluding remark
The levels of atorvastatin detected in waste water in this screening are comparable with previously reported levels. As atorvastatin was not detected in receiving waters, sediments or mussels, atorvastatin do not cause environmental concern.

Carbamazepine

Results from this study
The detected amounts of carbamazepine are presented graphically in Figure 7, trace ■. Carbamazepine was detected in all receiving waters in Oslofjord (10 - 20 ng/L), but in only one out of five samples from Tromsøsund (1 ng/L), and carbamazepine was not detected in sediment samples or mussels from Tromsøsund and Oslofjord. Carbamazepine was not analysed in the Ullevål and UNN effluent samples. Carbamazepine was detected in all STP effluent water samples in both Tromsø Breivika (250 - 400 ng/L) and VEAS (230 - 475 ng/L). Carbamazepine was also detected in sludge from VEAS (85 - 100 ng/g d.w.) and Breivika (120 - 195 ng/g d.w.).

Results from other studies

Carbamazepine has been detected at 9 - 1 100 ng/L in surface waters [8, 11], at 66 ng/L [12], at 7 - 251 ng/L [5], and at 0.7 ± 0.5 ng/L 5 km downstream of a STP [13] and at 30 - 1 100 ng/L in German rivers and streams [14]. In a previous Norwegian study, carbamazepine was not detected in receiving waters [15]. Carbamazepine has previously been detected in STP influent water at 290 - 400 ng/L [12], 2 593 ng/L [5], and 270 ng/L [15], and in effluent water at 80 - 80 000 ng/L [8, 11], at 50 - 6 300 ng/L [14], 3 117 ng/L [5], at 380 - 470 ng/L [12], and at 590 ± 125 ng/L [13]. Carbamazepine has been identified in sludge at concentrations up to 850 ng/g dw [16]. The removal efficiency is reported to be 0-55% [8, 12, 17]. In an Italian study the identified amount of carbamazepine was normalized to 28 mg/day/1000 inhabitants [17].

Ecotoxicological effects

An overview of known ecotoxicological effects of carbamazepine is presented in Table 16.

Life cycle and reproduction tests have been reported on the invertebrates, *Lumbriculus variegates* and *Chironomus riparius* and the endocrine disruption activity of carbamazepine has been suggested following the observation of inhibition of the formation of *Chironomus* pupae in the test [10].

In a French risk assessment, carbamazepine was prioritized due to a high PEC value, possible persistence in the aquatic environment and for being a CYP450 inducer [96]. Carbamazepine was included on a priority pollutants list for pharmaceuticals in Italy [17]. A risk quotient (PEC/PNEC) >1 calculated for carbamazepine suggests that there may be a risk to the water compartment [10].

Table 16: Ecotoxicological effects of carbamazepine.

Species	End point/effect	Concentration (ng/L)	Reference
Daphnia magna Crustacean	EC_{50}, 48 h	> 13 800 000	[10]
Ceriodaphnia dubia	EC_{50}, 48 h	77 000 000	[10]
Synechococcus leopolensis Cyanobacteria	EC_{50}	17 000 000	[18]
Cyclotella meneghiniana diatom	EC_{50}	10 000 000	[18]
Desmodesmus subspicatus green algae	EC_{50}	74 000 000	[18]
Pseudokirchneriella subcapitata green algae	EC_{50}	100 000 000	[18]
Danio rerio fish	EC_{50}	25 000 000	[18]
Chironomus riparius midge larva	EC_{50}	625 000	[18]
Lumbriculus variegates oligochaete worm	EC_{50}	10 000 000 ng/kg	[18]
Brachionus calyciflorus rotifer	EC_{50}	377 000	[18]
Ceriodaphnia dubia	EC_{50}	25 000	[18]
Onchorynchus mykiss rainbow trout	EC_{50} cytotoxicity to hepatocytes	118 000 000	[97]

Fate
The environmental photo degradation of carbamazepine is important and has been thoroughly studied [19]. The second order rate constant for photo degradation of carbamazepine is $9\pm1\times10^{9}$ M^{-1} s^{-1} [10]. Photolysis studies on carbamazepine indicate a very complex degradation pattern including formation of the very toxic compound acridine [19]. Carbamazepine is prevalent due to poor STP removal [11], with a 50% dissipation time of 82 ± 11 days [10]. Carbamazepine has an estimated environmental half-life of at least 80 days [3]. The substance must thus be regarded as potentially persistent.

Concluding remark
The levels of carbamazepine detected in this screening are comparable with previously reported levels, even though 50 times higher maximum concentrations have been reported. The detected maximum concentration of 20 ng/L in receiving water is less than three orders of magnitude less than the lowest reported effect concentration for carbamazepine. Thus, the detected concentrations of carbamazepine are of environmental concern.

Morphine
Results from this study
The detected amounts of morphine are presented graphically in Figure 7, trace ■. Morphine was detected in all five receiving water samples in Oslofjord (5 - 22 ng/L), but in no samples from Tromsøsund (<LoD 4 ng/L). Morphine was not detected in sediment samples (<LoD 6 - 10 ng/g d.w.) or mussels (<LoD 10-18 ng/g) from Tromsøsund and Oslofjord. Morphine was not analysed in the Ullevål and UNN effluent samples. Morphine was detected in all STP effluent water samples in both Tromsø Breivika (215 - 850 ng/L) and in three out of four samples from VEAS effluent water (530 - 1 000 ng/L). Morphine was not detected in sludge from VEAS (<LoD 9 ng/g d.w.) or Breivika (<LoD 8 ng/g d.w.).

Results from other studies
Morphine has previously been detected at 450-875 ng/L in Irish STP effluent water [20].

Ecotoxicological effects
No ecotoxicological effects of morphine are known.

Fate
The fate of morphine in the environment is unknown.

Concluding remark
The levels of morphine detected in STP effluent waters in this screening are comparable with previously reported levels. The highest detected concentration in receiving water was 22 ng/L. However, there are no reported ecotoxicological effects of morphine and therefore adverse environmental effects from morphine cannot be excluded and morphine is of some concern.

Naproxen
Results from this study
The detected amounts of naproxen presented graphically in Figure 7, trace ■. Naproxen was detected in all receiving water in both Tromsø (5 - 12 ng/L) and Oslofjord (24 - 54 ng/L). Naproxen was not detected in sediment samples or mussels from Tromsøsund and Oslofjord. Naproxen was not analysed in the Ullevål and UNN effluent samples. Naproxen was detected in all STP effluent water samples in both Tromsø Breivika (1200 - 3150 ng/L) and VEAS (60

- 1 100 ng/L). Naproxen was also detected in sludge from VEAS (10 - 11 ng/g d.w.) and Breivika (8 - 17 ng/g d.w.).

Results from other studies
Naproxen has been identified in surface waters at 10 - 800 ng/L [8], 9 - 21 ng/L [21], 12- 50 ng/L [5], and 10 - 390 ng/L rivers and streams [14]. Naproxen was detected in the Piteå River in the north of Sweden. The concentration varied between 0.46 - 1.2 ng/l depending on the distance from the STP [22]. Naproxen has been identified in various concentrations in STP influent water at 1 000 - 41 000 ng/L [8], 1 082 ng/L [5], 7 300 ng/L [21], and2 300 - 7 300 ng/L [22], and in effluent water at 100 - 60 000 ng/L [8], 1 700 ng/L [21], 50 - 520 ng/L [14], 400 ng/L [5], 3 200 - 3 400 ng/L [22], and 310 ± 150 ng/L [13]. For Naproxen, a STP removal efficiency of 40-100% [8], and 67% [23] has been reported. Naproxen has been detected in rainbow trout (*Oncorhynchus mykiss*) exposed to STP effluent water [24].

Ecotoxicological effects
Some known ecotoxicological effects of naproxen are presented in Table 17.

Table 17: Ecotoxicological effects of naproxen.

Species	End point/effect	Concentration (ng/L)	Reference
Daphnia magna crustacean	EC_{50} (immobilisation)	166 000 000	[10]
De. subspicatus green alga	EC_{50} (growth inhibition)	626 000 000	[10]
Ceriodaphnia dubia crustacean	EC_{50} (growth inhibition)	66 000 000	[10]
Water-flea (*Ceriodaphnia dubia*)	chronic probalistic NOEC 192 h	32 000	[23]
Th. platyrus, crustacean	LC_{50}	84 000 000	[10]
L. minor duck weed	EC_{50} (growth inhibition)	24 200 000	[10]

Naproxen has been tested on several species; however the chronic probalistic NOEC (192 h) of 32 000 ng/L on the water-flea *Ceriodaphnia dubia* [23] is a concentration three orders of magnitude less than empirical determined toxic values.
In a French risk assessment, naproxen was prioritised due to high PEC value and for showing renal toxicity [96].

Fate
Naproxen has no significant bioaccumulation potential (fass.se). Naproxen is susceptible to photo degradation in water [10]. The second order rate constant for photo degradation of naproxen is $9.6 \pm 0.5 \times 10^9$ M^{-1} s^{-1} [10]. The estimated half-life is 14 days [25].

Concluding remark
The levels of naproxen detected in this screening are lower or comparable with previously reported levels. The highest detected concentration of 54 ng/L in receiving water is less than three orders of magnitude less than the lowest reported effect concentration for naproxen. Thus, the detected concentrations of naproxen are of environmental concern.

Paracetamol
Results from this report
The detected amounts of paracetamol are presented graphically in Figure 7, trace ■. No paracetamol was detected in surface receiving water (<LoD 1 ng/L), sediment (<LoD 2 - 4 ng/g d.w.), or mussels (<LoD 15 ng/g w/w) in samples from Tromsøsund or Oslofjord. Paracetamol was detected in three out of four effluent water samples at VEAS (190 - 900

ng/L) and in all four samples from Breivika Tromsø (3 300 - 6 000 ng/L), however, paracetamol was not detected in sludge (<LoD 3 - 6 ng/g d.w.). Paracetamol was not analysed in Ullevål and UNN effluent.

Results from other studies
Paracetamol has previously been found at relatively high concentrations in surface water (up to 10 000 ng/L) [8, 26], however, paracetamol was not detected (<LoD 20 ng/L) in the river Tyne [27], or in German rivers and streams [14], but it was detected at 62 - 388 ng/L in River Taff [5]. Paracetamol has been analysed in sediment samples in one investigation the concentrations in sediments were 18 000 - 69 000 ng/kg dw. Paracetamol could not be detected in fish in this investigation albeit paracetamol was present in the river water phase at 110 - 360 ng/L [22]. Paracetamol was detected in STP influent water (VEAS) at 1 750 - 43 000 ng/L and in hospital effluent water (Ullevål) at 5 400 - 1 400 000 ng/L in a Norwegian study [28]. Another study reported paracetamol at 36 000 - 59 000 ng/L [22] in STP influent water. Paracetamol is reported to be 'efficiently removed' at STP [11], the removal was 98% in a German STP [14] and even a complete removal is reported [8]. Anyhow, paracetamol has previously been detected in STP effluent water at 80-7000 ng/L [8], 20 - 4 300 ng/L [28], 500 - 6 000 ng/L [14], 1 826 ng/L [5], 14 000 - 29 000 ng/L [29], and 12.6 ± 7.0 ng/L [13], but in the latter study it was not detected 5 km downstream of the STP.
Paracetamol has been identified in sludge in a concentration up to 1 400 000 ng/kg dw [16]. In a Norwegian study, no paracetamol was found in sludge [28].

Ecotoxicological effects
Some ecotoxicological effects of paracetamol are presented in Table 18. In a French risk assessment, paracetamol was prioritised due to high PEC value [96].

Table 18: Ecotoxicological effects of paracetamol.

Species	End point/effect	Concentration (ng/L)	Reference
Lemna gibba plant duckweed	EC_{50}	1 000 000	[18]
Hydra vulgaris cnidarians	EC_{50} of >10 000 ng/L		
Pimephales promelas Fathead minnow.	LC_{50} (96 h)	814 000 000	[37]
Daphnia magna	EC_{50} (immobilisation)	40 000 000	[11]

Fate
Paracetamol is slowly degraded in the aqueous environment (57% after 28 days), but does not bioaccumulate [7]. A log K_{ow} of 0.46 indicates that paracetamol is not expected to adsorb to suspended solids and sediment. But it is well documented that paracetamol released to surface water is rapidly transferred to the sediment despite a low K_{ow} and high pK_a (9.5). Most of the sediment-bound paracetamol has been proved to be involved in a strong binding (e.g. covalent binding) and could not be extracted by simple solvent extraction [98].

The annual consumption (2006) of the substance was approximately 170 tonnes in Norway. Paracetamol is not considered very toxic with a PNEC of 9.2 µg/L and is efficiently eliminated during sewage treatment processes when biological treatment is used [27, 28]. The removal efficiency for chemical/mechanical treatment is not known.

Concluding remark
Paracetamol was not detected in this screening in receiving waters, sediments or biota. It has occasionally been detected in surface waters in previous reports. Paracetamol do not cause environmental concern.

Propranolol
Results from this study
The amounts of propranolol detected are presented graphically in Figure 7, trace ■. Propranolol was detected in all receiving water samples collected from Oslofjord (1.6 - 3.0 ng/L) and in three out of five samples from Tromsøsund (0.5 - 1.2 ng/L), but propranolol was not detected in sediment samples or mussels from Tromsøsund and Oslofjord. Propranolol was not analysed in the Ullevål and UNN effluent samples. Propranolol was detected in all STP effluent water samples in both Tromsø Breivika (50 - 80 ng/L) and VEAS (22 - 42 ng/L). Propranolol was also detected in sludge from VEAS (23 - 30 ng/g d.w.) and Breivika (12 - 13 ng/g d.w.).

Results from other studies
Propranolol has been measured at 950 ng/L [11] and 10 - 850 ng/L [8] in surface water. Propranolol was detected at 35-107 ng/L in the river Tyne [27], and at 10 - 590 ng/L in German rivers and streams [14], and at 7 - 31 ng/L in River Taff [5]. In STP influent water, propranolol has been detected at 2 000 - 70 000 ng/L [8] 543 ng/L [5], and 20 ng/L [15]. Propranolol has been measured at a maximum concentration of 290 ng/L [11], 304 000 ng/L [8], 388 ng/L [5], and 10 ng/L [15], and 25 - 290 ng/L [14] in STP effluent water. The STP removal efficiency was reported to be 96% [11].

Ecotoxicological effects
An overview of ecotoxicological effects of propranolol is given in Table 19.

Table 19: Ecotoxicological effects of propranolol.

Species	End point/effect	Concentration (ng/L)	Reference
Daphnia magna	LOEC, growth	440 000	[26]
crustacean	LOEC, fecundity	110 000	[26]
	Lower heart rate	55 000	[26]
	EC_{50}, (immobilisation)	2 600 000	[11]
	mortality 48 h	2 000 000	[10]
Vibrio fischeri (bacterium)		81 000 000	[10]
Desmodesmus subspicatus green alga	growth rate 3 d	700 000	[10]
Pseudokirchneriella subcapitata green alga	growth inhibition 96 h	7 400 000	[10]
Ceriodaphnia dubia crustacean	(inhibition of mobility 48 h	1 000 000	[10]
Oryzias letipes	Mortality 48 h	25 000 000	[10]
fish	Fewer eggs released by fish 4-week exposure	500	[8]
Ceriodaphnia dubia fish	NOEC (reproduction)	125 000	[8]
	LOEC (reproduction)	250 000	
Hyalella azteca	reproduction (27 d)	100 000	[8]
Onchorynchus mykiss	cytotoxicity hepatocytes	25 900 000 000	[97]

Propranolol was found to be more toxic than the beta-blockers oxprenolol, atenolol, metoprolol, and nadolol [10]. Beta-adrenoceptors are 7-transmembrane receptor proteins coupled with different G-proteins that ultimately enhance the synthesis of the second

messenger signaling molecules cAMP. beta adrenoceptors have been identified in the fish *Oncorhynchus mykiss*, and in frog and turkey [8]. Propranolol is one of few pharmaceuticals that have been observed at environmental concentrations sufficiently high to cause effect in a non-target organism. In a French risk assessment, propranolol was prioritised due to high Ecotoxicological effects and for potential adverse effects on thyroids [96].

Fate
No information on the fate of propranolol has been found.

Concluding remark
The levels of propranolol detected in this screening are lower or comparable with previously reported levels. The detected concentration of 3 ng/L in receiving water is less than three orders of magnitude less than the lowest reported effect concentration for propranolol. Thus, the detected concentrations of propranolol are of environmental concern.

Sertraline
Results from this study
The detected amounts of sertraline are presented graphically in Figure 7, trace ■. Sertraline was not detected in any receiving water samples from Oslofjord or Tromsøsund (<LoD 2 ng/L), and sertraline was not detected in sediment samples (<LoD 1.5 - 4 ng/g d.w.) or mussels (<LoD 5 ng/g) from Tromsøsund and Oslofjord. Sertraline was not analysed in the Ullevål and UNN effluent samples. Sertraline was detected in all STP effluent water samples from VEAS (4 - 12 ng/L) and in three out of four samples from Tromsø Breivika (5 - 31 ng/L). Sertraline was detected in sludge from both VEAS (33 - 45 ng/g d.w.) and Breivika (13 ng/g d.w.).

Results from other studies
Sertraline was detected at 100 ng/L, 1.8-16.3 ng/L [31], in STP influent [15], and at 1 - 2 ng/L [31] in STP effluent water. In another study sertraline was found in STP effluent water at 4 - 15 ng/L in Tromsø and at 8 ng/L in Oslo, however, the compound was not detected in Longyearbyen STP effluent water [32]. The metabolite desmethyl-sertraline was also detected in some samples [32]. Sertraline is also one of few pharmaceuticals that have been detected in biota. Brooks et al. detected sertraline at 0.3 - 8 ng/g and its metabolite desmethyl sertraline at 0.5-30 ng/g (w/w) in muscle, liver and brain from the fish species *Ictalurus punctatus* (channel catfish), *Pomoxis nigromaculatus* (black crappie), and *Lepomis macrochirus* (bluegill) [33], living in the Pecan Creek in Texas, USA.

Ecotoxicological effects
Some known ecotoxicological effects of sertraline are presented in Table 20.

Table 20: Some known ecotoxicological concentrations of sertraline.

Species	End point/effect	Concentration (ng/L)	Reference
Ceriodaphnia dubia of to	LC_{50}	120 000	[99]
	EC_{50}	9 000	[18]
Lemna gibba duckweed	EC_{10}	1 000 000	[18]

Sertraline is toxic to algae and crustaceans in particular [3]. Fong demonstrated that SSRIs induced spawning in the zebra mussel *Dreissena polymorpha* (a non-target organism for SSRI) at sub μM concentrations [100], however, sertraline was not included in this study. In a

French risk assessment, sertraline was prioritised due to its serotoninergic activity, high Kow, high Ecotoxicological effects, and for being a CYP450 inhibitor [96].

Fate
9-32% of Sertraline remained after 45 days using an active sludge test. An environmental half life of 4.6 d was experimentally determined based on a modified EPA-TSCA (40CFR795.70) indirect photolysis protocol (fass.se). Monitoring data suggests that Sertraline should preferentially be monitored in sludge or sediment samples [101].

Concluding remark
The levels of sertraline detected in STP effluent waters in this screening are comparable with previously reported levels. However, sertraline was not detected in receiving waters, sediments or mussels and do not cause environmental concern.

Spiramycin
Results from this study
The detected amounts of spiramycin are presented graphically in Figure 7, trace ■. Spiramycin was not detected in any receiving water samples from Oslofjord or Tromsøsund (<LoD 3 ng/L), and spiramycin was not detected in sediment samples (<LoD 3 ng/g d.w.) or mussels (<LoD 5 ng/g) from Tromsøsund and Oslofjord. Spiramycin was not analysed in the Ullevål and UNN effluent samples. Spiramycin was detected in all STP effluent water samples from VEAS (9 - 30 ng/L), but not in any samples from Tromsø Breivika (<LoD 3 ng/L). Spiramycin was not detected in sludge from VEAS (<LoD 7 ng/g d.w.) and Breivika (<LoD 4 ng/g d.w.).

Results from other studies
Spiramycin was detected at 3 - 460 ng/L in river water [34].
A STP removal rate of 0% was reported for spiramycin [17].

Ecotoxicological effects
Some known ecotoxicological effects of spiramycin are given in Table 21. Spiramycin was included on a priority pollutants list for pharmaceuticals in Italy [17]. In this study, the identified amount of spiramycin was normalised to 35 mg/day/1000 inhabitants [17]. Antibiotics are commonly detected in the environment as contaminants. Exposure to antibiotics may induce antimicrobial resistance, as well as the horizontal gene transfer of resistance genes in bacterial populations. The multiple antibiotic resistance gene, *marA*, was found *Escherichia coli* and *Bacillus* species, the latter have not previously been reported to possess *marA*, in Italian river sediment and river waters by PCR measurement [34].

Table 21: Some known ecotoxicological effects of spiramycin.

Species	End point/effect	Concentration (ng/L)	Reference
Microcystis aeruginosa freshwater Cyanobacteria	EC_{50} (growth inhibition)	7 000	[35]
Selenastrum capricornutum green algae	EC_{50} (growth inhibition)	133 000	[35]

Fate
No information about the environmental fate of spiramycin was found, but the STP removal efficiency of 0% [17], suggests abiotic degradation to be more important than biotic.

Concluding remark
Spiramycin has previously only been detected in river waters, but in this report it was only detected in STP effluent waters. As spiramycin was not detected in receiving waters, sediments or biota and do not cause environmental concern.

Tamoxifen
Results from this study
The detected amounts of tamoxifen are presented graphically in Figure 7, trace ■. Tamoxifen was not detected in any receiving water samples from Oslofjord or Tromsøsund (<LoD 1 ng/L), and tamoxifen was not detected in sediment samples (<LoD 1 - 4 ng/g d.w.). However, tamoxifen was detected in one out of two mussels from Tromsøsund (5 ng/g), but in no mussels from Oslofjord (<LoD 5 - 10 ng/g). Tamoxifen was not analysed in the Ullevål and UNN effluent samples. Tamoxifen was not detected in any STP effluent water samples from VEAS and from Tromsø Breivika (<LoD 1 ng/L). Tamoxifen was detected in sludge from both VEAS (2 ng/g d.w.) and Breivika (1 ng/g d.w.).
Results from other studies
Tamoxifen has been detected at 70 - 250 ng/L in surface water [8], and at 25-210 ng/L in the river Tyne [27].
Tamoxifen was found at 150 ng/L in STP influent and at 10 - 400 ng/L in effluent water [8].
A STP removal efficiency of 0% has been reported [8].

Ecotoxicological effects
Tamoxifen is an important anti-estrogen acting by blocking the estrogen receptor. Data from partial life cycle and fish full life-cycle (FFLC) studies (maximum exposure periods of 42 and 211 d, respectively) support the overall conclusion that, for environmental risk assessment purposes, tamoxifen citrate has ^{adverse}NOEC and ^{adverse}LOEC concentrations of 5 120 and 5 600 ng/L, respectively [36].

Fate
No information about the fate of tamoxifen was found, but the STP removal efficiency of 0% [8], suggests abiotic degradation to be more important than biotic.

Concluding remark
Tamoxifen was not detected in receiving waters, sediment, STP effluent, or sludge, but it has previously been detected in river and STP effluent waters. However, tamoxifen was detected in one mussel sample and is therefore of some environmental concern.

Warfarin
Results from this study
The detected amounts of warfarin are presented graphically in Figure 7, trace ■. Warfarin was not detected in any receiving water samples from Oslofjord or Tromsøsund (<LoD 5 ng/L), and warfarin was not detected in sediment samples (<LoD 5 - 10 ng/g d.w.) or mussels (<LoD 15 - 25 ng/g) from Tromsøsund and Oslofjord. Warfarin was not analysed in the Ullevål or UNN effluent samples. Warfarin was detected in all STP effluent water samples from VEAS (10 - 70 ng/L) and from Tromsø Breivika (35 - 105 ng/L). Warfarin was detected in sludge from both VEAS (17 ng/g d.w.) and Breivika (10 -15 ng/g d.w.).

Results from other studies
Warfarin has been identified in sludge at concentrations up to 92 ng/g dw [16].

Ecotoxicological effects
The Ecotoxicological effects of warfarin is shown in Table 22.

Table 22: Ecotoxicological effects of warfarin.

Species	End point/effect	Concentration (ng/L)	Reference
Pseudokirchneriella subcapitata	EC_{50} 72 h	11 000 000	fass.se
Green alga	NOEC	2 500 000	
Daphnia magna	EC_{50} 48 h	111 000 000	fass.se
Water flea	NOEC	50 000 000	
Cyprinodon variegates	LC_{50} 96 h	497 000 000	fass.se
Fish	NOEC	250 000 000	

Fate
Biodegradation: 0% after 28 days (OECD 301D). Warfarin is potentially persistent (fass.se). Warfarin has a log Kow of 2.70 and a water solubility of 17 mg/L, which indicate that Warfarin is expected to adsorb to suspended solids and sediment; however, the potential for bio concentration in aquatic organisms is low. Warfarin hydrolyses very slowly in water with a half-life (pH 7, 25°C) of 16 years [37].

Concluding remark
The levels of warfarin detected in sludge in this screening are comparable with previously reported levels. Warfarin was not detected in receiving waters, sediments or mussels and do not cause environmental concern.

5.3 Selected hospital human pharmaceuticals

5.3.1 Antibiotics

Amoxicillin
Results from this study
Amoxicillin was not detected in any receiving water samples from Oslofjord or Tromsøsund (<LoD 15 - 100 ng/L), and amoxicillin was not detected in sediment samples (<LoD 2 - 5 ng/g d.w.) or mussels from Tromsøsund and Oslofjord (<LoD 12 - 20 ng/g). Amoxicillin was not detected in Ullevål or UNN effluent samples (<LoD 20 - 200 ng/L). Amoxicillin was not detected in STP effluent water samples from VEAS (<LoD 7 - 17 ng/L) and Tromsø Breivika (<LoD 25 - 175 ng/L). Amoxicillin was not detected in sludge from VEAS (<LoD 20 - 35 ng/g d.w.), and Breivika (LoD 145 - 230 ng/g d.w.).

Results from other studies
Amoxicillin has not previously been detected in environmental samples.

Ecotoxicological effects
Strains of *Escherichia coli* were isolated from an STP and resistance to amoxicillin was observed in three isolates [10]. In a Korean risk assessment, the hazard classification of amoxicillin was ‘very high to aquatic organisms’ [103]. In a British risk assessment, amoxicillin was judged to have a high potential to reach the environment, high usage, high

toxicity profile classification, resulting in a high priority for detailed risk assessment [104]. In a French risk assessment, amoxicillin was prioritized due to high PEC value and for being an antibiotic agent [96]. Amoxicillin has been included on a priority pollutants list for pharmaceuticals in Italy [17]. Amoxicillin was identified in 2003 as one of 56 aquaculture medicines that have a high potential of entering the environment [105]. Some known ecotoxicological effects of amoxicillin are presented in Table 23.

Table 23: Selected ecotoxicological effects of amoxicillin.

Species	End point/effect	Concentration (ng/L)	Reference
Hydra vulgaris invertebrate cnidarians	EC_{50}	10 000	[18]
Lemna gibba plant duckweed	EC_{10}	1 000 000	[18]
Ps. subcapitata green algae		Nontoxic	[38]
Cyclotella meneghiniana phytoplankton		Nontoxic	[38]
Synechococcus leopolensis cyanobacterium	EC_{50}	2 220	[38]
Selenastrum capricornutum	IC_{50}72 h	630 000 000	fass.se
Green alga	NOEC	530 000 000	
Microcystis aeruginosa	EC_{50} 7 days	3 700	fass.se
Bluegreen alga			
Synechococcus leopolensis	EC_{50} (growth inhibition) 96 h	2 220	fass.se
Bluegreen alga			
Daphnia magna	EC_{50} 48 h	>2 300 000 000	fass.se
	NOEC	2 300 000 000	
Lepomis macrochirus	EC_{50} 96 h	>930 000 000	fass.se
Bluegill sunfish	NOEC	930 000 000	
Oncorhynchus mykiss	EC_{50} 96 h	>1 000 000 000	fass.se
Rainbow trout	NOEC	1 000 000 000	fass.se
	Hepatocytes	Nontoxic	[97]
Microcystis aeruginosa freshwater cyanobacteria	EC_{50} (growth inhibition)	3 700	[102]
Selenastrum capricornutum freshwater green alga,	NOEC	250 000 000	[102]
Rhodomonas salina marine cryptophycean	EC_{50} (growth inhibition)	3 108 000 000	[102]

Fate

Amoxicillin has a hydrolytic half-life of 50-113 days at pH 7 (OECD 111) and a photolytic half-life of 1.13 days at pH 7.5 [3]. Amoxicillin does not bioaccumulate with a log P = 0.87 (fass.se).

Concluding remark

Amoxicillin has not previously been detected in environmental samples. Amoxicillin was not detected in any sample in this screening and do not cause environmental concern.

Cefotaxime

Results from this study

The detected amounts of cefotaxime are presented graphically in Figure 8, trace ■. Cefotaxime was not detected in any receiving water samples from Oslofjord or Tromsøsund (<LoD 1 - 2 ng/L), and cefotaxime was not detected in sediment samples (<LoD 0.2 - 0.8 ng/g d.w.) or mussels from Tromsøsund and Oslofjord (<LoD 0.5 - 1.4 ng/g). Cefotaxime was

detected in all Ullevål effluent samples (30 - 440 ng/L) and in three out of four UNN effluent samples (60 - 325 ng/L). Cefotaxime was detected in all STP effluent water samples from VEAS (35 - 55 ng/L) and Tromsø Breivika (110 - 580 ng/L). Cefotaxime was not detected in sludge from VEAS and Breivika (<LoD 3 - 5 ng/g d.w.).

Results from other studies
Cefotaxime has not previously been detected in environmental samples.

Ecotoxicological effects
Cefotaxime has a reported toxicity to Zebra fish *Danio rerio* (EC_{50} 96 h) of > 500 000 000 ng/L [3]. Genotoxicity testing showed negative results (internal report). Cefotaxime is not teratogenic (fass.se). Ash et al carried out a study on water samples taken from streams in USA and found evidence of bacterial resistance to e.g. cefotaxime [39].

Fate
Cefotaxime is potentially persistent with a 13% degradation in 28 days, but the substance is light sensitive [3]. Furthermore, it is unlikely to bioaccumulate in aquatic organisms based on its solubility (550 000 mg/L).

Concluding remark
Cefotaxime has not previously been detected in environmental samples. Cefotaxime was not detected in receiving waters, sediments or biota. Cefotaxime do not cause environmental concern.

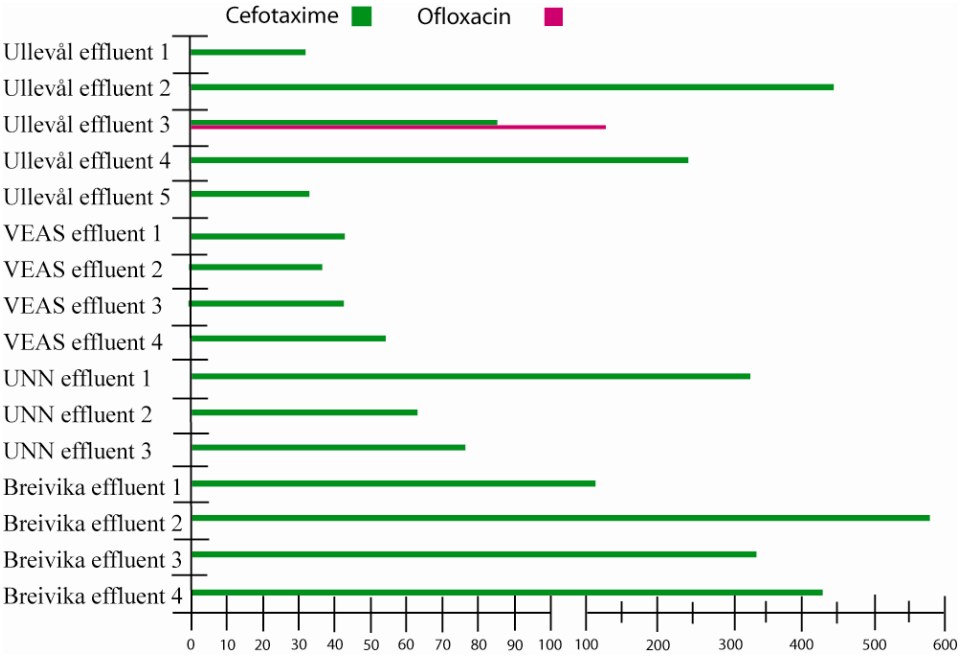

Figure 8: The figure shows the measured concentrations of cefotaxime (trace ■), and ofloxacin, (trace ■). The concentrations are presented as ng/L for aqueous samples and as ng/g (dw) for solid samples.

Cefalotin

Results from this study
Cefalotin was not detected in any receiving water samples from Oslofjord or Tromsøsund (<LoD 3 - 7 ng/L), and cefalotin was not detected in sediment samples (<LoD 1 - 3 ng/g d.w.) or mussels from Tromsøsund and Oslofjord (<LoD 2 - 4 ng/g). Cefalotin was not detected in

Ullevål effluent samples (<LoD 7 - 33 ng/L) and in UNN effluent samples (<LoD 80 - 210 ng/L). Cefalotin was not detected in any STP effluent water samples from VEAS (<LoD 2 - 5 ng/L) and Tromsø Breivika (<LoD 80 - 160 ng/L). Cefalotin was not detected in sludge from VEAS and Breivika (<LoD 9 - 14 ng/g d.w.).

Results from other studies
Cefalotin has not previously been detected in environmental samples.

Ecotoxicological effects
No data on the Ecotoxicological effects of cefalotin has been found.

Fate
The information regarding the environmental fate of cefalotin is scarce.

Concluding remark
Cefalotin has not previously been detected in environmental samples and was not detected in this screening. Cefalotin do not cause environmental concern.

Meropenem
Results from this study
Meropenem was not detected in any receiving water samples from Oslofjord or Tromsøsund (<LoD 3 - 30 ng/L), and meropenem was not detected in sediment samples (<LoD 0.6 - 2.2 ng/g d.w.) or mussels from Tromsøsund and Oslofjord (<LoD 5 - 10 ng/g). Meropenem was not detected in Ullevål effluent samples (<LoD 7 - 70 ng/L) and in UNN effluent samples (<LoD 55 - 100 ng/L). Meropenem was not detected in any STP effluent water samples from VEAS (<LoD 2 - 5 ng/L) and Tromsø Breivika (<LoD 40 - 100 ng/L). Meropenem was not detected in sludge from VEAS (<LoD 8 - 13 ng/g d.w.) and Breivika (<LoD 60 - 90 ng/g d.w.).

Results from other studies
Meropenem has not been detected in environmental samples.

Ecotoxicological effects
The available ecotoxicological data is scarce. Meropenem has a reported EC_{50} (48 h) of >900 000 000 ng/L to *Daphnia magna* [3].

Fate
Meropenem is not rapidly biologically degraded, but it is prone to undergo hydrolysis with reported half lives of 63 h (pH 7) and 12 min (pH 9). Meropenem does not bioaccumulate due to a log P <0,001.

Concluding remark
Meropenem was not detected in any sample in this screening and do not cause environmental concern.

Ofloxacin

Results from this study

The detected amounts of ofloxacin are presented graphically in Figure 8, trace ■. Ofloxacin was not detected in any receiving water samples from Oslofjord or Tromsøsund (<LoD 1 - 2 ng/L), and ofloxacin was not detected in sediment samples (<LoD 0.2-0.8 ng/g d.w.) or mussels from Tromsøsund and Oslofjord (<LoD 0.6-1.6 ng/g). Ofloxacin was detected in one out of four Ullevål effluent samples (129 ng/L), but not in UNN effluent samples (<LoD 25 - 50 ng/L). Ofloxacin was not detected in any STP effluent water samples from VEAS (<LoD 1 - 2 ng/L) and Tromsø Breivika (<LoD 20 - 45 ng/L). Ofloxacin was not detected in sludge from VEAS and Breivika (<LoD 6 - 7 ng/g d.w.).

Results from other studies

Ofloxacin was detected at 20 - 300 ng/L in Italian river water [34]. Ofloxacin has been detected in Finland at 30-150 ng/L in STP influent water, whereas concentrations up to 10 ng/L was detected in the STP effluent [12]. The related compound ciprofloxacin has been measured at 3-87 µg/L in Swiss hospital wastewater [40].

A STP removal rate of 57% was reported for ofloxacin [17]. In this study, the identified amount of ofloxacin was normalized to 233 mg/day/1000 inhabitants [17].

Ecotoxicological effects

In a French risk assessment, ofloxacin was prioritized due to a high PEC value, ATB, and for having high Ecotoxicological effects towards cyanobacteria [96, 106]. Ofloxacin was included on a priority pollutants list for pharmaceuticals in Italy [17]. Ofloxacin was calculated to have an acceptable risk (PEC/PNEC < 1) [10]. Information available to date does not suggest any endocrine disrupting potential (fass.se).

Some known ecotoxicological effects of ofloxacin are presented in Table 24.

Table 24: Some known ecotoxicological effects of ofloxacin.

Species	End point/effect	Concentration (ng/L)	Reference
Daphnia magna	EC_{50} (48 h)	76 600 000	fass.se
Microcystis aeruginosa	EC_{50}	21 000	[107]
Lemna minor	EC_{50}	126 000	[107]
Pseudokirchneriella subcapitata green alga	EC_{50}	12 100 000 2 500 000	[107] [18]
Synechococcus leopolensis cyanobacterium	EC_{50}	5 000	[18]
Cyclotella meneghiniana diatom	EC_{50}	30 000	[18]
Brachionus calyciflorus rotifer	EC_{50}	12 500 000	[18]
Ceriodaphnia dubia water flea	EC_{50}	10 000 000	[18]
Lemna gibba duckweed	EC_{50}	120 000	[18]
Vibrio fischeri marine bacterium	EC_{50}	14 000	[108]

Fate

Strongly adsorbs to soil and is highly active in hospital wastewaters [11, 40]. The medicine shows no biodegradation, but the substance is light sensitive, with a photo degradation half-life of 0.3 - 10.6 days. The second order rate constant for photo degradation of ofloxacin is ~5×10^{9} M^{-1} s^{-1} [10].

Concluding remark
The levels of ofloxacin detected in hospital effluent water in this screening are comparable with previously reported levels. Ofloxacin was not detected in receiving waters, sediments or biota and this do not cause environmental concern.

Penicillin G
Results from this study
Penicillin G was not detected in any receiving water samples from Oslofjord or Tromsøsund (<LoD 2.5 - 7 ng/L), and penicillin G was not detected in sediment samples (<LoD 0.5 - 2.5 ng/g d.w.) or mussels from Tromsøsund and Oslofjord (<LoD 2 - 5 ng/g). Penicillin G was not detected in Ullevål or UNN effluent samples (<LoD 40 - 200 ng/L). Penicillin G was not detected in any STP effluent water samples from VEAS (<LoD 14 - 19 ng/L) and Tromsø Breivika (<LoD 65 - 110 ng/L). Penicillin G was not detected in sludge from VEAS and Breivika (<LoD 8 - 12 ng/g d.w.).

Results from other studies
Penicillin G has not been detected in environmental samples.

Ecotoxicological effects
Some known ecotoxicological effects of penicillin G are presented in Table 25.

Table 25: Some ecotoxicological effects of penicillin G.

Species	End point/effect	Concentration (ng/L)	Reference
Microcystis aeruginosa freshwater Cyanobacteria	EC_{50}	6 000	[35]
Selenastrum capricornutum green algae	NOEC	100 000 000	[35]

Fate
It was observed that penicillin G was unstable due to hydrolysis and photolysis [35]

Concluding remark
Penicillin G was not detected in any sample in this screening and does not cause environmental concern.

Pivmecillinam
Results from this study
Pivmecillinam was not detected in any receiving water samples from Oslofjord or Tromsøsund (<LoD 0.2 - 0.5 ng/L), and pivmecillinam was not detected in sediment samples (<LoD 0.1 - 0.3 ng/g d.w.) or mussels from Tromsøsund and Oslofjord (<LoD 0.3 - 0.7 ng/g). Pivmecillinam was not detected in Ullevål or UNN effluent samples (<LoD 1 - 11 ng/L). Pivmecillinam was not detected in any STP effluent water samples from VEAS (<LoD 0.3 - 0.6 ng/L) and Tromsø Breivika (<LoD 4 - 11 ng/L). Pivmecillinam was not detected in sludge from VEAS and Breivika (<LoD 1 - 2 ng/g d.w.).

Results from other studies
Pivmecillinam has not been detected in environmental samples.

Ecotoxicological effects
No ecotoxicological data are currently available.

Fate
The fate of pivmecillinam in the environment is unknown.

Concluding remark
Pivmecillinam was absent in all samples in this careening and do not cause environmental concern.

5.3.2 X-ray contrast agents

Iodixanol, iohexol, and iopromide
Results from this study

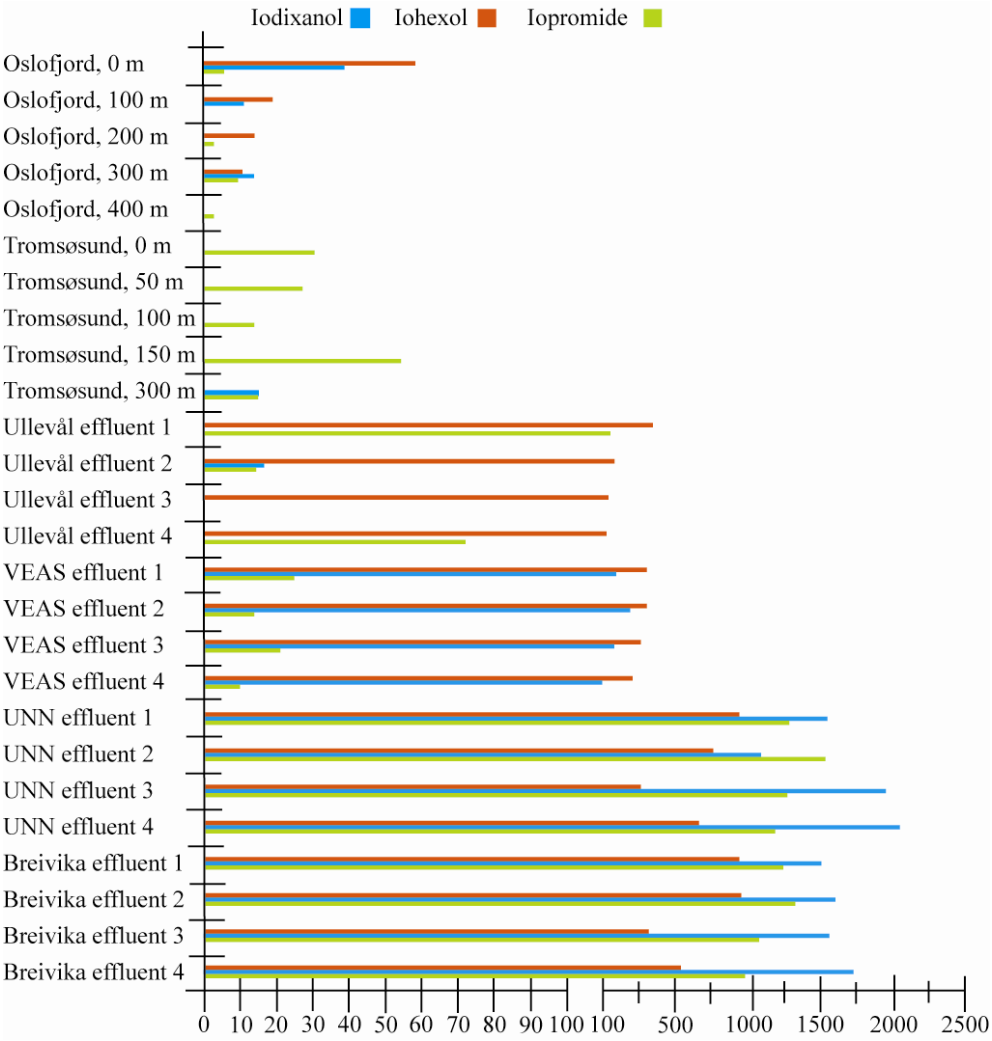

Figure 9: The figure shows the measured concentrations of iohexol (trace ■), iodixanol, (trace ■), and iopromide (trace ■).The concentrations are presented as ng/L for aqueous samples and as ng/g (dw) for solid samples.

The detected amounts of iodixanol in aqueous samples in this study are presented graphically in Figure 9, trace ■. Iodixanol was detected in three out of five receiving water samples from Oslofjord (10 - 40 ng/L) and one out of five samples from Tromsøsund (14 ng/L). Iodixanol was detected at 7 ng/g dw and at 5 - 7 ng/g dw in Oslofjord and Tromsøsund sediment, respectively. Iodixanol was not analysed in mussels from Tromsøsund and Oslofjord. Iodixanol was detected in one out of four Ullevål effluent samples (16 ng/L) and in all UNN effluent samples (1 200 - 2 100 ng/L). Iodixanol was detected in all STP effluent water

samples from VEAS (100 - 200 ng/L) and Tromsø Breivika (1 500 - 1 750 ng/L). Iodixanol was detected in sludge from VEAS (10 ng/g d.w.), but not in sludge from Breivika (<LoD 1 ng/g d.w.).

The detected amounts of iohexol in aqueous samples are presented graphically in Figure 9, trace ■. Iohexol was detected in four out of five receiving water samples from Oslofjord (10 - 60 ng/L), but not in any samples from Tromsøsund (<LoD 20 ng/L). Iodixanol was detected in sediment samples Oslofjord (13 ng/g d.w.), but not from Tromsøsund (<LoD 0.8 ng/g d.w.). Iodixanol was not analysed in mussels from Tromsøsund and Oslofjord. Iohexol was detected in all Ullevål effluent samples (120 - 330 ng/L) and UNN effluent samples (250 - 890 ng/L). Iohexol was detected in all STP effluent water samples from VEAS (220 - 310 ng/L) and Tromsø Breivika (340 - 920 ng/L). Iodixanol was not detected in sludge from VEAS and Breivika (<LoD 0.8 ng/g d.w.).

The detected amounts of iopromide in aqueous samples are presented graphically in Figure 9, trace ■. Iopromide was detected in four out of five receiving water samples from Oslofjord (3 - 9 ng/L), and in all samples from Tromsøsund (10 - 50 ng/L). Iopromide was detected in sediment samples from Tromsøsund and Oslofjord at 1 - 2 and 1 ng/g dw, respectively. Iopromide was not analysed in mussels from Tromsøsund and Oslofjord. Iopromide was detected in three out of four Ullevål effluent samples (13-150 ng/L) and UNN effluent samples (1 200 - 1 525 ng/L). Iopromide was detected in all STP effluent water samples from VEAS (7-24 ng/L) and Tromsø Breivika (960 - 1 360 ng/L). Iopromide was not detected in sludge from VEAS and Breivika (<LoD 0.5 ng/g d.w.).

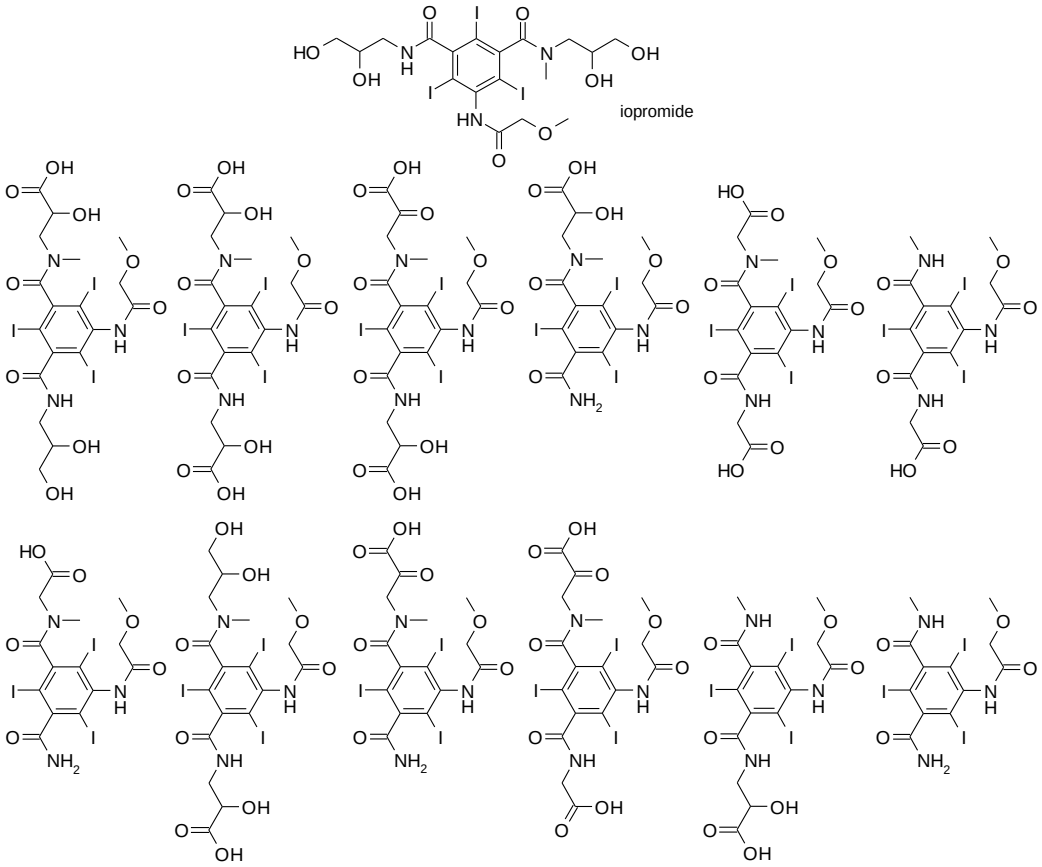

Figure 10: Iopromide and its transformation products [42].

Results from other studies
Iopromide was detected at 50 - 11 000 ng/L in German STP effluent water with no effective removal in the STPs [41]. It has been measured at relatively high concentrations, i.e. up to 11 000 ng/L in municipal STP effluents [11]. No reports on the detection of iohexol and iodixanol in the environment were found.

Ecotoxicological effects
The toxicity of the metabolites of iopromide are unknown [11]. Iopromide is toxic towards a (unspecified) cyanobacterium with an EC_{50} of 68 000 000 ng/L. It has also been tested to the invertebrate *Daphnia magna*, yielding an EC_{50} of >1 000 000 000 ng/L [18]. No reports on the Ecotoxicological effects of iohexol and iodixanol were found.

Fate
Iopromide is very resistant to biodegradation and extremely persistent [11]. Iopromide and twelve (bio)transformation products (see Figure 10) were detected in STP effluent water and concentrations up to 3.7 ± 0.9 µg/L [42]. The environmental effect(s) of the transformation products has not been assessed [42]. No reports on the fate of iohexol and iodixanol in the environment were found.

Concluding remark
The levels of iopromide detected in this screening are comparable with previously reported levels. Iodixanol and iohexol has not previously been detected in environmental samples. The maximum detected concentration of 40 ng/L (iodixanol), 60 ng/L (iohexol), and 50 ng/L (iopromide) in receiving waters is more than five orders of magnitude less than the lowest reported effect concentration for the compounds. Thus, the detected concentrations of iodixanol, iohexol, and iopromide do not cause environmental concern.

5.3.3 Cytostatics

Bortezomib
Results from this study
Bortezomib was not detected in any receiving water samples from Oslofjord and Tromsøsund (<LoD 10 ng/L), and bortezomib was not detected in sediment samples (<LoD 25 ng/g d.w.). Mussels were not analysed for their bortezomib content. Bortezomib was not detected in Ullevål or UNN effluent samples (<LoD 90 - 500 ng/L). Bortezomib was not detected in any STP effluent water samples from VEAS or Tromsø Breivika (<LoD 15 - 275 ng/L). Bortezomib was not detected in sludge from VEAS or Breivika (<LoD 1200 ng/g d.w.).

Results from other studies
Bortezomib has not been detected in environmental samples.

Ecotoxicological effects
Some known ecotoxicological effects bortezomib are presented in Table 26.

Table 26: Some known ecotoxicological effects of bortezomib.

Species	End point/effect	Concentration (ng/L)	Reference
Scenedesmus subspicatus	EC_{50} (72 h)	300 000	[3]
Green alga	NOEC	100 000	[3]
Daphnia magna	EC_{50} (48 h)	450 000	[3]
Water-flea,	NOEC	170 000	[3]
Brachydanio rerio	LC_{50} (96 h)	1 100 000	[3]
Zebra fish	NOEC	460 000	[3]

Fate
No information is available on degradation and bioaccumulation of bortezomib.

Concluding remark
Bortezomib do not cause environmental concern as it was absent in all samples.

Docetaxel
Results from this study
Docetaxel was not detected in any receiving water samples from Oslofjord or Tromsøsund (<LoD 1 ng/L), and docetaxel was not detected in sediment samples (<LoD 40 ng/g d.w.). Mussels were not analysed for their docetaxel content. Docetaxel was not detected in Ullevål and UNN effluent samples (<LoD 5 - 30 ng/L). Docetaxel was not detected in any STP effluent water samples from VEAS and Tromsø Breivika (<LoD 2 - 8 ng/L). Docetaxel was not detected in sludge from VEAS and Breivika (<LoD 500 ng/g d.w.).

Results from other studies
Docetaxel has not been detected in environmental samples.

Ecotoxicological effects
For docetaxel, the EC_{50} (48 h) is 3 700 000 ng/L for *Daphnia magna* and the EC_{50} (72 h) is 545 000 ng/L for the algae *Scenedesmus subspicatus*.

Fate
Docetaxel is slowly degraded with a hydrolytic half-life at pH 7 of 28 days. Bioaccumulation of docetaxel cannot be excluded [3].

Concluding remark
Docetaxel was absent in all samples and therefore it do not cause environmental concern.

Doxorubicin and doxorubicinol

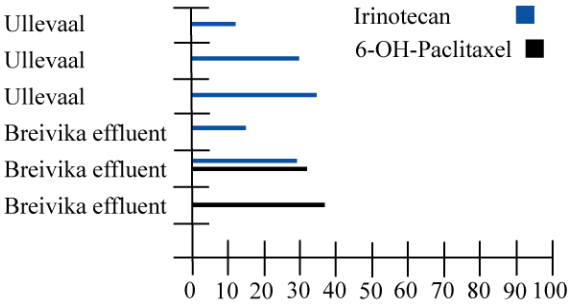

Figure 11: The figure shows the measured concentrations of irinotecan (trace ■) and 6-OH-paclitaxel (trace ■). The concentrations are presented as ng/L.

Results from this study
Doxorubicin was not detected in any receiving water samples from Oslofjord or Tromsøsund (<LoD 1 ng/L), and doxorubicin was not detected in sediment samples (<LoD 85 ng/g d.w.). Mussels were not analysed for their doxorubicin content. Doxorubicin was not detected in Ullevål and UNN effluent samples (<LoD 4 - 9 ng/L). Doxorubicin was not detected in any STP effluent water samples from VEAS and Tromsø Breivika (<LoD 1 - 8 ng/L). Doxorubicin was detected in sludge from VEAS and Breivika (1 450 - 5 600 ng/g d.w.). Doxorubicinol was not detected in any sample, and its LoD is in the same order of magnitude as doxorubicin.

Results from other studies
Doxorubicin has been detected at 500 ng/L in hospital effluent water [43, 44].

Ecotoxicological effects
Doxorubicin is toxic to *Daphnia magna*, with a reported toxic concentration (EC_{50}) of 9 900 000 ng/L [3].

Fate
No data are available on the degradation and bioaccumulation on doxorubicin and doxorubicinol.

Concluding remark
Doxorubicin and its metabolite doxorubicinol were not detected in any sample and they do not cause environmental concern.

Irinotecan
Results from this study
The detected amounts of irinotecan are presented graphically in Figure 11, trace ■. Irinotecan was not detected in any receiving water samples from Oslofjord or Tromsøsund (<LoD 1 - 4 ng/L), and irinotecan was not detected in sediment samples (<LoD 750 ng/g d.w.). Mussels were not analysed for their irinotecan content. Irinotecan was detected in three out of four Ullevål effluent (15 - 35 ng/L), but in none of the UNN effluent samples (LoD 1 ng/L). Irinotecan was not detected in any STP effluent water samples from VEAS (<LoD 0.8 ng/L), but in two out of four effluent samples from Tromsø Breivika (15 - 30 ng/L). Irinotecan was not detected in sludge from VEAS and Breivika (<LoD 1 100 ng/g d.w.).

Results from other studies
Irinotecan has not previously been detected in the environment.

Ecotoxicological effects
No ecotoxicological data are available for irinotecan.

Fate
The fate of irinotecan in the environment is not known.

Concluding remark
Irinotecan has previously not been detected in environmental samples, but it was detected in hospital and STP effluent water in this screening. However, irinotecan was not detected in receiving waters, sediments or biota and do not cause environmental concern.

Paclitaxel and 6-OH-Paclitaxel

Results from this study

Paclitaxel was not detected in any receiving water samples from Oslofjord or Tromsøsund (<LoD 1 - 4 ng/L), and paclitaxel was not detected in sediment samples (<LoD 20 ng/g d.w.). Mussels were not analysed for their paclitaxel content. Paclitaxel was not detected in Ullevål or UNN effluent (<LoD 3 - 6 ng/L). Paclitaxel was not detected in any STP effluent water samples from VEAS or Breivika (LoD 1 - 6 ng/g). Paclitaxel was not detected in sludge from Breivika or VEAS. (<LoD 300 ng/g d.w.).

The detected amounts of 6-OH-paclitaxel are presented graphically in Figure 11, trace ■. 6-OH-Paclitaxel was not detected in any receiving water samples from Oslofjord and Tromsøsund (<LoD 1 - 6 ng/L), and 6-OH-paclitaxel was not detected in sediment samples (<LoD 45 ng/g d.w.). Mussels were not analysed for their 6-OH-paclitaxel content. 6-OH-Paclitaxel was not detected in Ullevål or UNN effluent (<LoD 4 - 9 ng/L). 6-OH-Paclitaxel was not detected in any STP effluent water samples from VEAS (<LoD 3 -14 ng/L), but in two out of four effluent samples from Tromsø Breivika (35 - 40 ng/L). 6-OH-Paclitaxel was not detected in sludge from VEAS and Breivika (<LoD 650 ng/g d.w.).

Results from other studies

Paclitaxel and 6-OH-paclitaxel have not been found in environmental samples.

Ecotoxicological effects

For Paclitaxel, a NOEC of 740 000 ng/L is reported for *Daphnia magna* [3].

Fate

Paclitaxel has a log Kow of 3.5 (pH 7), however, the bioaccumulation potential to organisms is low based on metabolism and biodegradation data. Paclitaxel is readily biodegraded as it exhibited 68.1% mineralization to $^{14}CO_2$ in the first 14 days of a biodegradation study [3].

Concluding remark

Paclitaxel and its metabolite 6-OH-paclitaxel have not previously been detected in environmental samples. In this screening, paclitaxel was not detected in any sample, but the metabolite was detected in STP effluent waters, but not in receiving waters, sediments or biota. Therefore, irinotecan and its metabolite 6-OH-irinotecan do not cause environmental concern.

5.4 Selected aquaculture medicines

5.4.1 Aquaculture medicines

Cypermethrin and Deltamethrin

Results from this study

Cypermethrin was not detected in any water samples from neither Fish farm 1 nor Fish farm 2 (<LoD 2 ng/L). Cypermethrin was not detected in sediment samples from Fish farm 1 or Fish farm 2 (<LoD 5 ng/g d.w.). Mussels from Fish farm 1 and Fish farm 2 did not contain cypermethrin (<LoD 5 ng/g w/w).

Deltamethrin was not detected in any water samples from neither Fish farm 1 nor Fish farm 2 (<LoD 10 ng/L). Deltamethrin was not detected in sediment samples from Fish farm 1 or Fish

farm 2 (<LoD 15 ng/g d.w.). Mussels from Fish farm 1 and Fish farm 2 did not contain deltamethrin (<LoD 15 ng/g w/w).

Results from other studies
Cypermethrin was detected at 2 - 5 ng/g (dw) in river sediments and at 6 - 66 ng/g (dw) in drain mouths, and deltamethrin at the same sites at 2 - 5 ng/g (dw) and 13 - 78 ng/g (dw) [45]. In river water cypermethrin was detected at 10 ng/L (dry season) and 9 - 26 ng/L (wet season). Deltamethrin was not detected in the dry season, but were detected at 4 ng/L in the wet season [45].

Ecotoxicological effects
The pyrethroid insecticides have been reported to be present in sediments at concentrations exceeding toxicity thresholds for sensitive invertebrates, and testing with the amphipod *Hyalella azteca* has commonly shown acute toxicity [45]. *Hyalella azteca* survival rate varied from 9% in sediment containing 2.2 ng/g (dw) cypermethrin and no deltamethrin, to 70% in a sediment containing 4.7 ng/g (dw) and no deltamethrin. For comparison, a sediment containing 2.4 and 5.1 ng/g dw of cypermethrin and deltamethrin, respectively, gave a *H. azteca* survival of 45% [45].

An EC_{50} of >39 900 000 ng/L for deltamethrin exposed to *Vibrio fischeri* was reported by Hernando et al. [46].
In a British risk assessment, deltamethrin was judged to have a high potential to reach the environment, unknown usage, medium toxicity profile classification, resulting in a medium priority for detailed risk assessment, whereas cypermethrin was judged to have a high potential to reach the environment, medium usage, medium toxicity profile classification, resulting in a low priority for detailed risk assessment [104]. Deltamethrin and cypermethrin were identified in 2003 as two of 56 aquaculture medicines that have a high potential of entering the environment [105].

Fate
The fate of cypermethrin and deltamethrin in the environment is scarcely described, but it is suggestive from the Ecotoxicological effects investigations described above that the compounds will adsorb to solids.

Concluding remark
Cypermethrin and deltamethrin were not detected in any samples in this screening, but they have previously been detected in environmental samples. Due to their absence, cypermethrin and deltamethrin do not cause environmental concern.

Emamectin
Results from this study
The detected amounts of emamectin are presented graphically in Figure 12, trace ■. Emamectin was not detected in any water samples from neither Fish farm 1 nor Fish farm 2 (<LoD 1 ng/L). Emamectin was detected in two out of five sediment samples from Fish farm 1 (2.3 - 2.4 ng/g d.w.), and in three out of five sediment samples from Fish farm 2 (2.1 - 6.5 ng/g d.w.). Mussels from Fish farm 1 and Fish farm 2 did not contain emamectin (<LoD 2 ng/g w/w).

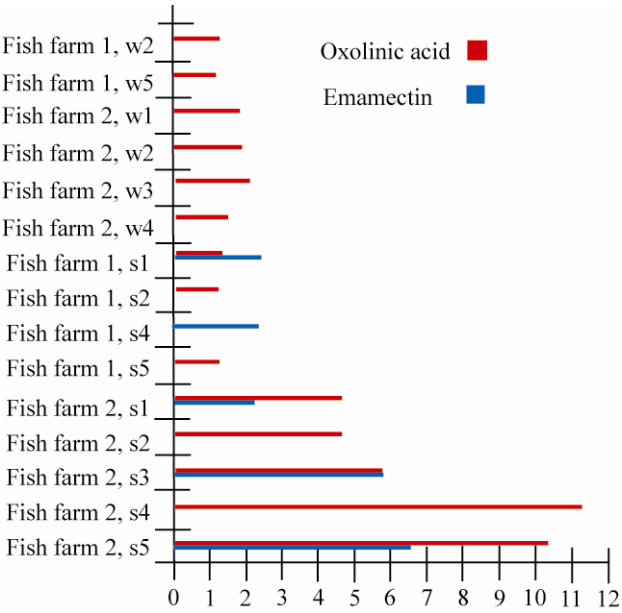

Figure 12: The figure shows the measured concentrations of oxolinic acid (trace ■) and emamectin, (trace ■). The concentrations are presented as ng/L for aqueous samples and as ng/g (dw) for solid samples..

Results from other studies

Emamectin has not previously been detected in environmental samples.

Ecotoxicological effects

In a British risk assessment, emamectin was judged to have a high potential to reach the environment, unknown usage, medium toxicity profile classification, resulting in a medium priority for detailed risk assessment [104]. Emamectin was identified in 2003 as one of 56 aquaculture medicines that have a high potential of entering the environment [105]. There has been reported evidence for field evolved resistance to emamectin in *Spodoptera litura* (Fabricius), a serious pest causing enormous losses to important cultivated crops, such as cotton and soybean [109]. Some known ecotoxicological effects of emamectin are presented in Table 27.

Table 27. Some ecotoxicological effects of emamectin.

Species	End point/effect	Concentration (ng/L)	Reference
Vibrio fischeri	EC_{50}	6 300 000	[46]

Fate

The fate emamectin in the environment is not known.

Concluding remark

Emamectin has not previously been detected in environmental samples. The maximum detected concentration of 6.5 ng/g in the sediment is more than five orders of magnitude less than the lowest reported effect concentration for emamectin. Thus, the detected concentrations of emamectin do not cause environmental concern.

Fenbendazole

Results from this study

Fenbendazole was not detected in any water samples from neither Fish farm 1 nor Fish farm 2 (<LoD 2 ng/L). Fenbendazole was not detected in sediment samples from Fish farm 1 and

Fish farm 2 (<LoD 3 ng/g d.w.). Mussels from Fish farm 1 and Fish farm 2 did not contain fenbendazole (<LoD 3 ng/g w/w).

Results from other studies
Fenbendazole has not been detected in environmental samples.

Ecotoxicological effects
In a Korean risk assessment, the hazard classification of fenbendazole was 'very high to aquatic organisms' [103]. In a British risk assessment, fenbendazole was judged to have a unknown potential to reach the environment, medium usage, low toxicity profile classification, resulting in a very low priority for detailed risk assessment [104]. An EC_{50}-48 h of 16 500 ng/L of fenbendazole to *Daphnia magna* is reported [47]. Fenbendazole is toxic to both crustaceans and fish, PNEC 10 ng/L, depending on the assessment factor used [3].

Fate
The fate of fenbendazole in the environment is unknown.

Concluding remark
Fenbendazole has not previously been detected in environmental samples and was also not detected in this screening. The absence of fenbendazole in all samples does not cause environmental concern.

Flumequine
Results from this study
Flumequine was not detected in any water samples from neither Fish farm 1 nor Fish farm 2 (<LoD 1 ng/L). Flumequine was not detected in sediment samples from Fish farm 1 or Fish farm 2 (<LoD 1 ng/g d.w.). Mussels from Fish farm 1 and Fish farm 2 did not contain flumequine (<LoD 1 ng/g w/w).

Results from other studies
Flumequine has not been detected in environmental samples.

Ecotoxicological effects
Some known ecotoxicological effects of flumequine are shown in Table 28.

Table 28: Some known ecotoxicological concentrations of flumequine.

Species	End point/effect	Concentration (ng/L)	Reference
Microcystis aeruginosa.	EC_{50}	1 960 000	[107]
Pseudokirchneriella subcapitata	EC_{50}	5 000 000	[107]
	EC_{50}	8 500 000	[110]
Vibrio fischeri	EC_{50}	198 000	[46]
	EC_{50}	40 000 000	[110]

Fate
Information about the fate of flumequine in the environment is scarce.

Concluding remark
Flumequine has not previously been detected in environmental samples and was also not detected in this screening. The absence of flumequine in all samples does not cause environmental concern.

Oxolinic acid

Results from this study

The detected amounts of oxolinic acid are presented graphically in Figure 12, trace ■. Oxolinic acid was detected in two out of five water samples from Fish farm 1 (1.1 - 1.2 ng/L) and in four out of five water samples from Fish farm 2 (1.7 - 2.1 ng/L). At Fish farm 2, samples were taken at 0, 50 m, 100 m, 300 m, and 500 m from the plant. The latter sample did not contain oxolinic acid. Oxolinic acid was detected in three out of five sediment samples from Fish farm 1 (1.2 - 1.3 ng/g d.w.) and in all sediment samples from Fish farm 2 (5 - 11 ng/g d.w.). Mussels from Fish farm 1 and Fish farm 2 did not contain oxolinic acid (<LoD 2 ng/g w/w).

Results from other studies

Oxolinic acid has been detected in shrimp at 0.3-4.0 ng/g [48].

Ecotoxicological effects

Some known ecotoxicological effects of oxolinic acid are presented in Table 29.

Table 29: Ecotoxicological effects of oxolinic acid.

Species	End point/effect	Concentration (ng/L)	Reference
Vibrio fischeri	EC_{50}	198 000	[46]
	EC_{50}	150 000 000	[110]
Mytilus edulis blue mussel	No bioaccumulation		[111]
Pseudokirchneriella subcapitata	EC_{50}	37 000 000	[110]

Oxolinic acid was identified in 2003 as one of 56 aquaculture medicines that have a high potential of entering the environment [105].

Fate

No information about the fate of oxolinic acid in the environment is available.

Concluding remark

Oxolinic acid has previously been detected in biota, but was only detected in receiving water and sediment samples in this screening and not in mussel. The detected concentration of 11 ng/g in sediments is more than four orders of magnitude less than the lowest reported effect concentration for oxolinic acid. Thus, the detected concentrations of oxolinic acid do not cause environmental concern.

Praziquantel

Results from this study

Praziquantel was not detected in any water samples from neither Fish farm 1 nor Fish farm 2 (<LoD 3 ng/L). Praziquantel was not detected in sediment samples from Fish farm 1 or Fish farm 2 (<LoD 3 ng/g d.w.). Mussels from Fish farm 1 and Fish farm 2 did not contain praziquantel (<LoD 3 ng/g w/w).

Results from other studies

Praziquantel has not been detected in environmental samples.

Ecotoxicological effects
Praziquantel has a NOEL for vertebrates at 20 mg/kg/day [49]. Praziquantel was determined to have a NOEC of >1000 mg/kg dung to the larvae of the dung beetle *Aphodius constans* [49].

Fate
The fate of praziquantel in the environment is not known.

Concluding remark
Praziquantel has not previously been detected in environmental samples and was also not detected in this screening. The absence of praziquantel in all samples does not cause environmental concern.

5.4.2 Comment on the aquaculture medicines detected in the fish farms

In fish farm 1 emamectin benzoate was used in the period June 30 - July 6, 2008, and deltamethrin was used throughout the whole year from January 7. In fish farm 2 oxolinic acid was used in the period July 11 - 21. Deltamethrin was not detected in any sample. Emamectin was not detected in surface water, but was detected in the sediment at both fish farms. The concentrations were slightly higher at fish farm 2, where there had been no reported use in 2008, than at fish farm 1. Oxolinic acid was detected in surface at a distance of 300 m from fish farm 2, but not at 500 m. At fish farm 1, oxolinic acid was detected in two samples; 50 m and 500 m from the farm at a concentration just above the LoD. For the sediment, oxolinic acid was detected just above LoD at fish farm 1, whereas ten times higher concentrations were detected at fish farm 2.

5.5 Selected personal care products

Avobenzone
Results from this study
Avobenzone was not detected in receiving samples (<LoD 2 ng/L). Avobenzone was not detected in sediment samples from Oslofjord and Tromsøsund (<LoD 5 ng/g). Mussels from Oslofjord and Tromsøsund did not contain avobenzone (<LoD 5 ng/g (w/w)). Avobenzone was not analysed in Ullevål or UNN effluent samples. In STP effluent water, avobenzone was not detected in samples from VEAS and Breivika (LoD 2 ng/L). Avobenzone was not detected in sludge from Breivika and VEAS (<LoD 20 ng/g d.w.).

Results from other studies
Avobenzone has previously been found in swimming pools and in trace amounts <LoD (20 ng/L)-24 ng/L in surface water [52-54]. Avobenzone was not detected in surface water in Swiss Lakes (<2 ng/L) [51]. It was also not detected in lakes with inputs from recreational activities such as swimming and bathing [112]. The compound was not detected in a recent Norwegian screening [113].

Ecotoxicological effects
Avobenzone showed no endocrine disrupting activity when tested for estrogenic activity (MCF-7 cells) or anti-androgenic activity (MDA-kb2 cells) [55]. However, it has been shown that other UV-filters, i.e. 3-benzylidene camphor and 4-methylbenzylidene camphor, disrupt the androgen and estrogen balance in laboratory rats and their progeny [55, 114]. Avobenzone

showed no estrogenic activity on rainbow trout estrogenic receptor (rtER) and human ER (hER) [56].

Fate
Water solubility of 1.52 mg/L, log Kow 2.41 [54]. Avobenzone degrades in sunlight (www.smartskincare.com).

Concluding remark
Avobenzone was not detected in any sample in this screening and has previously only been detected at very low concentrations. The absence of avobenzone in all samples does not cause environmental concern.

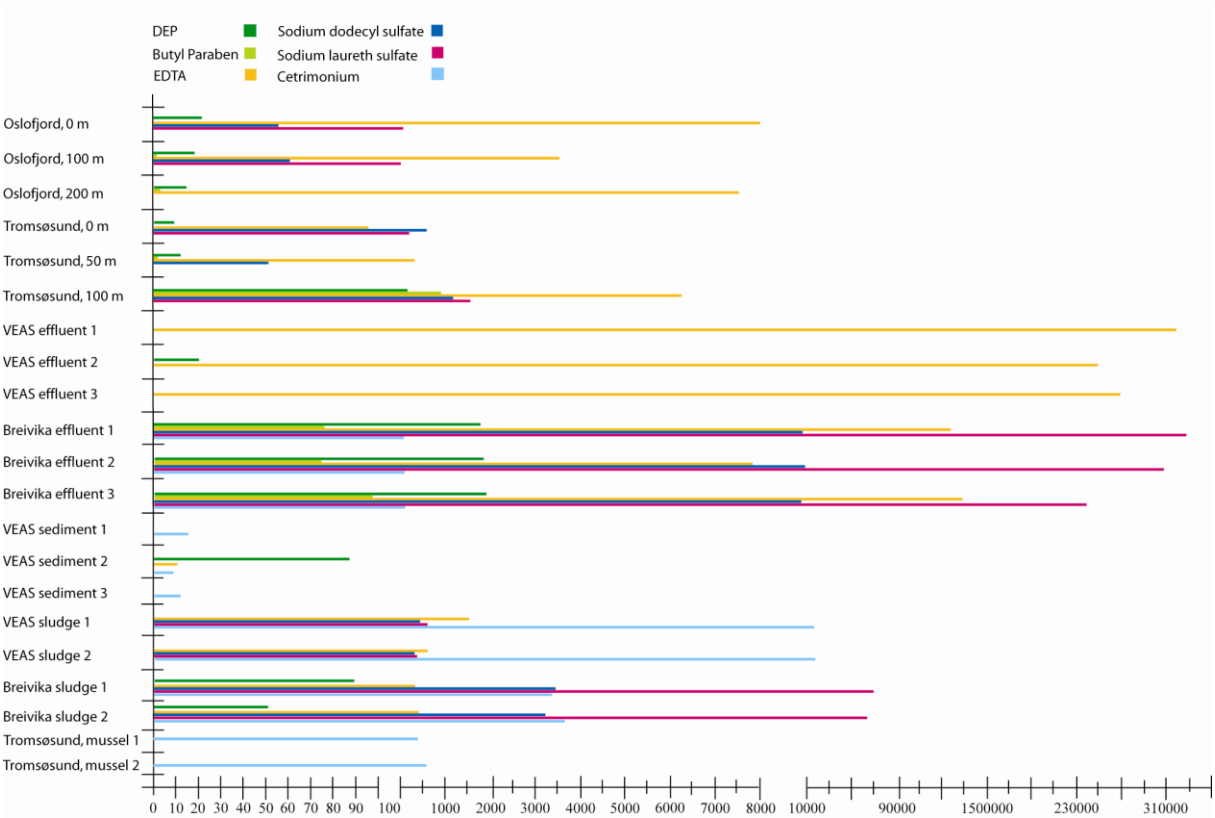

Figure 13: The figure shows the measured concentrations of diethyl phthalate (DEP; trace ■), butyl paraben (trace ■), EDTA (trace ■), dodecyl sulfate (trace ■), and laureth sulfate (trace ■). The concentrations are presented as ng/L for aqueous samples and as ng/g (dw) for solid samples.

Butyl paraben

Results from this study
The detected amounts of butyl paraben are presented graphically in Figure 13, trace ■. Butyl paraben was detected at 2, and 4 ng/L at a distance of 100 and 200 m, respectively, from the VEAS outlet in Oslofjord. Similarly in Tromsøsund the equidistant samples gave butyl paraben concentrations of 3 and 900 ng/L, respectively. Butyl paraben was not detected in sediment samples from Oslofjord and Tromsøsund (<LoD 4 ng/g). Mussels from Oslofjord and Tromsøsund did not contain butyl paraben (<LoD 4 ng/g w/w). Butyl paraben was not analysed in Ullevål or UNN effluent samples. In STP effluent water, butyl paraben was not detected in samples from VEAS (LoD 2 ng/g), but in three out of four effluent samples from Tromsø Breivika (77 - 97 ng/L). Butyl paraben was not detected in sludge from Breivika and VEAS (<LoD 4 ng/g d.w.).

Results from other studies
Butyl paraben was detected in STP influent at 45 ng/L [57], and 52 ng/L [5]. Butyl paraben has been detected in STP effluent water at 10 - 260 ng/L [58], and 100 ng/L [57]. In sludge, butyl paraben was detected at 63 ng/g dw [57]. A removal efficiency of 96% for butyl paraben in a WWTP was observed [58].

Ecotoxicological effects
An excellent review of toxic effects of parabens on humans is written by Dabre and Harvey (2008) [59]. Parabens are weak estrogens [58, 59]. The anti-androgenergic effect of butyl paraben was investigated [60], and it inhibited testosterone induced transcriptional activity by 19% at 1 940 000 ng/L.

Fate
Butyl paraben was shown to be highly stable against photo degradation, but it was readily biodegradable with half-times varying between 9.5 and 16 h [58]. Butyl paraben has a log D_{ow} = 3.43 (pH 7), suggesting particle sorption to be important in the environment [58].

Concluding remark
The levels of butyl paraben detected in this screening are higher than or comparable with previously reported levels. The detected concentration of 900 ng/L in receiving water is three to four orders of magnitude less than the lowest reported effect concentration for butyl paraben. Thus, the detected concentrations of butyl paraben cause some environmental concern.

Cetrimonium salt
Results from this study
The detected amounts of cetrimonium are presented graphically in Figure 13, trace ■. Cetrimonium was not detected in the receiving waters of Oslofjord and Tromsøsund (<LoD 40 ng/L). Cetrimonium was detected in sediment samples from Oslofjord (8 - 17 ng/g d.w.), but not in sediments from Tromsøsund (<LoD 4 ng/g d.w.). Mussels from Oslofjord (9.5 ng/g w/w) and Tromsøsund (400 - 500 ng/g d.w.) contained cetrimonium. Cetrimonium was not analysed in Ullevål or UNN effluent samples. In STP effluent water, cetrimonium was not detected in samples from VEAS (<LoD 40 ng/L), but in samples from Tromsø Breivika (3 100 - 3 600 ng/L). Cetrimonium was detected in sludge from Breivika (3 300 - 3 600 ng/g d.w.) and VEAS (12 000 - 15 000 ng/g d.w.).

Results from other studies
Cetrimonium has been detected at a median concentration of 160 - 8 400 µg/kg dw in sludge, and at concentrations between 1.8 and 120 µg/kg in Austrian river sediments [61].

Ecotoxicological effects
Some known ecotoxicological effects of cetrimonium are presented in Table 30.

Table 30: Some known ecotoxicological effects of cetrimonium.

Species	End point/effect	Concentration (ng/L)	Reference
Echinogammarus tibaldii, Crustacean	LC_{50},	160 000	[115]
Spirodela oligorhiza Duckweed	EC_{50}, growth inhibition	18 000 000	[116]
Microcystis sp. Phytoplankton	EC_{50}, growth inhibition	25 000	[62]

Fate
The fate of cetrimonium in the environment is not known.

Concluding remark
The levels of cetrimonium detected in sludge and sediment samples in this screening are comparable with previously reported levels. However, the highest concentrations of cetrimonium were detected in mussels at up to 500 ng/g, which are two orders of magnitude less than the lowest reported effect concentration for cetrimonium. Thus, the detected concentrations of cetrimonium are of environmental concern.

Cocoamidopropyl betaine
Results from this study
Cocoamidopropyl betaine was not detected in the receiving waters of Oslofjord or Tromsøsund (<LoD 10 ng/L). Cocoamidopropyl betaine was not detected in sediment samples from Oslofjord or Tromsøsund (<LoD 20 ng/g). Mussels from Oslofjord and Tromsøsund were not analysed for their cocoamido propyl betaine content. Cocoamidopropyl betaine was not analysed in Ullevål or UNN effluent samples. In STP effluent water, cocoamidopropyl betaine was not detected in samples from Tromsø Breivika (<LoD 200 ng/L) or from VEAS (<LoD 50 ng/L). Cocoamidopropyl betaine was detected in sludge from Breivika (1 500 - 1 700 ng/g d.w.) and VEAS (72-73 ng/g d.w.).

Results from other studies
Cocoamidopropyl betaine has not previously been monitored.

Ecotoxicological effects
Some known ecotoxicological effects of cocoamidopropyl betaine are presented in Table 31.

Table 31: Ecotoxicological effects of cocoamidopropyl betaine.

Species	End point/effect	Concentration (ng/L)	Reference
Pseudokirchneriella subcapitata alga	EC_{50}	1 500 000 ± 600 000	[63]
Scenedesmus subspicatus alga	EC_{50}	740 000 ± 60 000	[63]
Phaeodactylum tricornutum diatom	EC_{50}	410 000 ± 50 000	[63]
Skeletonema costatum diatom	EC_{50}	260 000 ± 30 000	[63]

Fate
The alkyl chain may undergo β- or ω-oxidation (see lauryl/laureth sulfate below).

Concluding remark
Cocoamidopropyl betaine has not previously been detected in environmental samples, and it was not detected in receiving water, sediment or mussel samples of this screening, but in sludge at 1 700 ng/g. Thus, the absence of cocoamidopropyl betaine in receiving samples does not cause environmental concern.

Diethylphthalate (DEP)
Results from this study
The detected amounts of DEP in this study are presented graphically in Figure 13, trace ■.
DEP was detected at 22, 19, and 14 ng/L at a distance of 0, 100, and 200 m, respectively, from the VEAS outlet in Oslofjord. Similarly in Tromsøsund the equidistant samples gave DEP concentrations of 17, 12, and 139 ng/L, respectively. DEP was detected in a sediment

sample from Oslofjord (87 ng/g d.w.), but DEP was not detected in Tromsøsund sediments (<LoD 20 ng/g). A mussel from Oslofjord had a DEP content of 9 ng/g (w/w), but no DEP was found in other mussels. DEP was not analysed in Ullevål or UNN effluent samples. In STP effluent water, DEP was detected in one out of four samples from VEAS (21 ng/g), and in three out of four effluent samples from Tromsø Breivika (1 775 - 1 935 ng/L). DEP was detected in sludge from Breivika (50 - 90 ng/g d.w.), but not in sludge from VEAS (<LoD 20 - 50 ng/g d.w.).

Results from other studies
DEP was detected in French rivers at 80 - 420 ng/L, which is comparable to concentrations measured in other European rivers and surface water [64], and in a Swedish river at 20 - 130 ng/L [65]. Phthalates have been identified in all environmental compartments [67]. DEP was detected in low concentrations in surface sediments in Swedish reference lakes (<1 900 - 36 000 ng/kg d.w.). In sediments from urban areas was the concentrations somewhat higher (<1 900 - 79 000 ng/kg d.w.) [66].

Ecotoxicological effects
Phthalates have been shown to be endocrine disruptors, that is, they are weak estrogen mimics. The suspected "gender bender" properties for DEP have been thoroughly described [68]. In a study from India, infertile men had significantly higher DEP concentration in their semen than fertile men [69]. A high DEP semen concentration also had a higher proportion of cells with depolarized mitochondria and a higher sperm cell content of reactive oxygen species (ROS) [69].Estrogen mimicking activity was observed in *Cyprinus carpio* at concentrations of 96 000 ng/L, which is 500 times lower than the LC_{50} of the same species [67].

Fate
DEP has log K_{ow} 2.38, a water solubility of 1 100 000 ng/L, and a vapour pressure (25°C) of ~$5 \cdot 10^{-4}$ mmHg [70]. The aqueous hydrolysis half-life of DEP is 8.8 yr, whereas the atmospheric half life is 1.8-18 days [70]. In soil, 90% of inoculated DEP was degraded within a week [70].

Concluding remark
The levels of DEP detected in this screening are comparable with previously reported levels. The detected concentration of 140 ng/L in receiving water is less than three orders of magnitude less than the lowest reported effect concentration for DEP. Furthermore, DEP was detected both in sediment and biota samples. Thus, the detected concentrations of DEP and its presence in biota is of environmental concern.

EDTA
Results from this study
The detected amounts of EDTA in this study are presented graphically in Figure 5, trace ■. EDTA was detected at 7 900, 3 700, and 7 600 ng/L at a distance of 0, 100, and 200 m, respectively, from the VEAS outlet in Oslofjord. Similarly in Tromsøsund the equidistant samples gave EDTA concentrations of 100, 200, and 6 000 ng/L, respectively. EDTA was not detected in sediment samples from Oslofjord and Tromsøsund (<LoD 10 ng/g). Mussels from Oslofjord and Tromsøsund did not contain EDTA (<LoD 15 ng/g w/w). EDTA was not analysed in Ullevål or UNN effluent samples. In STP effluent water, EDTA was detected in three out of four samples both from VEAS (240 000 - 310 000 ng/L) and Tromsø Breivika

(79 000 - 130 000 ng/L). EDTA was detected in sludge from Breivika 280 - 390 ng/g d.w.) and VEAS (600 - 1 100 ng/g d.w.).

Results from other studies
EDTA was detected in water samples from Lake Vättern (Sweden) at 5 000 - 7 000 ng/L [72]. EDTA was measured along a gradient at a Swedish pulp and paper factory. Close to the discharge point was the concentration in the surface water 200 000 ng/L. At a distance of 10 km was the concentration 30 000 ng/L [72].

Ecotoxicological effects
One possible mechanism for EDTA Ecotoxicological effects is through enhanced uptake of undesired metal cations. The reaction between EDTA (Z^{m-}) and a metal ion (Me^{n+}) is:

$$Me^{n+} + Z^{m-} \leftrightarrow MeZ^{(m-n)-}$$

The equilibrium constant for this reaction is given by:
$K_{MeZ} = [MeZ^{(m-n)-}]/([Me^{n+}][Z^{m-}])$,
where $[MeZ^{(m-n)-}]$ is the concentration of the metal-EDTA complex, $[Me^{n+}]$ is the concentration of the metal ion, and $[Z^{m-}]$ is the concentration of the $EDTA^{4-}$ ion [71]. Some K_{MeZ} values are given in Table 32.

Table 32: Equilibrium constants between $EDTA^4$ and selected metal cations . [117].

Metal ion	Ag^{+}	Mg^{2+}	Ca^{2+}	Sr^{2+}	Ba^{2+}	Mn^{2+}	Fe^{2+}	Co^{2+}	Ni^{2+}
Log K_{MeZ}	7.3	8.7	10.7	8.6	7.8	13.8	14.3	16.3	18.6

Metal ion	Cu^{2+}	Zn^{2+}	Cd^{2+}	Hg^{2+}	Pb^{2+}	Al^{3+}	Fe^{3+}	V^{3+}	Th^{4+}
Log K_{MeZ}	18.8	16.5	16.5	21.8	18.0	16.1	25.1	25.9	23.2

All cations present in the environment may compete for EDTA binding. Although EDTA itself is non-toxic to mammals at environmental relevant concentrations, there is a concern that EDTA has the potential to perturb the natural speciation of metals, and to influence metal bioavailability [71]. Furthermore, the proper function of many enzymes is dependent on metal cations as co-factors. The high concentrations of EDTA may lead to the remobilization of toxic metals from sediments to aquifers, consequently posing a risk to groundwater drinking water [71]. A LD_{50} of 24 000 000 ng/L was reported for bluegill (*Lepomis macrochirus*) [73].

Fate
EDTA is only slowly biodegradable, and therefore is rather persistent in the environment [71, 74]. An important sink for EDTA in the environment is photo degradation but is only valid for the Fe-EDTA complex [72, 75-77]. EDTA may be degraded under special conditions in the activated sludge in STP [78, 79].
EDTA has a low affinity to particulate matter is therefore not expected to be associated to the sediments [74, 118, 119].

Concluding remark
The levels of EDTA detected in this screening are comparable with previously reported levels. The maximum detected concentration of 7 900 ng/L in receiving water is three to four orders of magnitude less than the lowest reported effect concentration for EDTA. Thus, the detected concentrations of EDTA are of some environmental concern.

Sodium dodecyl sulfate (SDS) or sodium lauryl sulfate

Results from this study

The detected amounts of SDS in this study are presented graphically in Figure 13, trace ■. SDS was detected at 55, 60, and <40 ng/L (LoD) at a distance of 0, 100, and 200 m, respectively, from the VEAS outlet in Oslofjord. Similarly in Tromsøsund the equidistant samples gave SDS concentrations of 500, <40 (LoD), and 1 100 ng/L, respectively. SDS was not detected in sediment samples from Oslofjord and Tromsøsund (<LoD 40 ng/g). Mussels from Oslofjord and Tromsøsund were not analysed for their SDS content. SDS was not analysed in Ullevål or UNN effluent samples. In STP effluent water, SDS was detected in three out of four samples from Tromsø Breivika (9 600 -10 000 ng/L). At VEAS, the SDS content was less than 300 ng/L (LoD). SDS was detected in sludge from Breivika (3 200 - 3 400 ng/g d.w.) and VEAS (350 - 490 ng/g d.w.).

Results from other studies

SDS has not previously been analysed in environmental samples.

Ecotoxicological effects

Some known ecotoxicological effects of SDS are presented in Table 33.

Table 33: Some known ecotoxicological effects of SDS.

Species	End point/effect	Concentration (ng/L)	Reference
Vibrio fischeri bacteria	EC_{50}	8 200 000	[120]
Pseudomonas putida bacteria	EC_{50}	>150 000 000	[120]
Pseudokirchneriella subcapitata alga	EC_{50}	3 100 000 ± 500 000	[63]
Scenedesmus subspicatus algae;	EC_{50}	400 000 ± 60 000	[63]
Phaeodactylum tricornutum diatoms	EC_{50}	900 000 ± 40 000	[63]
Skeletonema costatum diatoms	EC_{50}	360 000 ± 40 000	[63]

The hypothesized potentiating effect of combining an- and cationic surfactants was not observed [120].

Fate

SDS is less soluble in cold water than sodium laureth sulfate [80]. SDS may undergo β-oxidation mediated by Pseudomonas sp. [81, 82].

Concluding remark

The detected maximum concentration of 1 100 ng/L of SDS in receiving water is less than three orders of magnitude less than the lowest reported effect concentration for SDS, but SDS is known to metabolize fast in the environment (β-oxidation). Thus, the detected concentrations of SDS are of some environmental concern only.

Sodium laureth sulfate

Results from this study

The detected amounts of laureth sulfate (SDSEO) are presented graphically in Figure 13, trace ■. Laureth sulfate was detected at 110, 110, and <40 ng/L (LoD) at a distance of 0, 100, and 200 m, respectively, from the VEAS outlet in Oslofjord. Similarly in Tromsøsund the equidistant samples gave laureth sulfate concentrations of 210, <40 (LoD), and 1 600 ng/L, respectively. Laureth sulfate was not detected in sediment samples from Oslofjord and Tromsøsund (<LoD 80 ng/g). Mussels from Oslofjord and Tromsøsund were not analysed for their laureth sulfate content. Laureth sulfate was not analysed in Ullevål or UNN effluent

samples. In STP effluent water, laureth sulfate was detected in three out of four samples from Tromsø Breivika (230 000 - 320 000 ng/L). At VEAS, the laureth sulfate content was less than 600 ng/L (LoD). Laureth sulfate was detected in sludge from Breivika (58 000 - 60 000 ng/g d.w.) and VEAS (370 - 520 ng/g d.w.).

Results from other studies
Sodium laureth sulfate has not been detected in other environmental samples.

Ecotoxicological effects
Some ecotoxicological effects of sodium laureth sulfate are presented in Table 34.

Table 34. Some ecotoxicological effects of sodium laureth sulfate.

Species	End point/effect	Concentration (ng/L)	Reference
Pseudokirchneriella subcapitata alga	EC_{50}	3 500 000 ± 700 000	[63]
Scenedesmus subspicatus algae;	EC_{50}	500 000 ± 50 000	[63]
Phaeodactylum tricornutum diatoms	EC_{50}	500 000 ± 70 000	[63]
Skeletonema costatum diatoms	EC_{50}	370 000 ± 80 000	[63]

Fate
The detergent (sodium) lauryl ether sulfate may undergo ω-oxidation, see **Error! Reference source not found.** [83].

Concluding remark
The highest detected concentration of 1 600 ng/L of SDSEO in receiving water is less than three orders of magnitude less than the lowest reported effect concentration for SDSEO. However, SDSEO is known to undergo ω-oxidation and the detected concentrations are of some environmental concern.

5.6 Influence of Northern environmental conditions

Comparable samples have been collected from greater Oslo and Tromsø. As explained below different degradation rates for the target compounds might be expected. However, the dataset in this study is extremely small and differences in sewage treatment are severe. It was therefore currently not possible to identify such a north-south difference.

On the other hand, in the published reports on the environmental fate of pharmaceuticals, the experiments have been conducted at warmer and lighter conditions than in the Norwegian environment. Chemical reaction rates decrease with decreasing temperatures and the amount of photons (sun light) reaching the Earth's surface decreases with increased latitude [123]. As an example, one metabolite of ibuprofen, carboxylated ibuprofen, seems to be significantly more stable in the cold seawater environment around Tromsø (annual average temperature 4 - 6 °C) compared to middle latitude environments [124]. Consequently, this renders the Northern environments more vulnerable for negative effects from the discharge of pharmaceuticals. A more thorough monitoring of the receiving waters, sediment and biota for pharmaceuticals and selected metabolites, is thus recommended, as there is a steady increase in the amount of pharmaceuticals purchased and thus a similar increase in the amount released into the environment.

6. Conclusions

In the discussion on the *environmental concerns* with the identified pharmaceuticals in the present study, the following criteria were been applied:

(i) If the compound was not detected or only detected in waste water, the compound was assessed to be of no or little environmental concern.

(ii) For compounds detected in receiving water and/or sediment, its highest detected concentration was compared with the worst case Ecotoxicological effects concentration found in the scientific literature:
   a. If the difference between highest observed concentration and the worst case Ecotoxicological effects concentration found in the scientific literature was more than 100 000, the compound was assessed to be of *little or no environmental concern*.
   b. If the difference between highest observed concentration and the worst case Ecotoxicological effects concentration found in the scientific literature was more than 1 000, but less than 100 000, the compound was assessed to be of *some environmental concern.*
   c. If the difference between highest observed concentration and the worst case Ecotoxicological effects concentration found in the scientific literature was less than 1000, the compound was assessed to be of *environmental concern*. 1000 was chosen as a safety factor as this often is applied as a safety factor in environmental risk assessments

(iii) Compounds identified in biota are automatically of *environmental concern*.

Figure 14 presents the highest determined concentrations in this study in the different matrices along with the lowest ecotoxicological concentration reported for the compounds in question. Compounds determined to be of little or no environmental concern are shaded grey. Compounds of some environmental concern are shaded yellow whereas a red shading is used for compounds that are present in biota or receiving waters at concentrations sufficiently high (relative to their known ecotoxicological effects) to be of environmental concern.

Pharmaceuticals

Amoxicillin, bortezomib, cefalotin, docetaxel, doxorubicin, doxorubicinol, meropenem, paclitaxel, and *penicillin G* were not detected and will not be further discussed. *Amitriptyline, atorvastatin, sertraline*, and *warfarin* were detected in STP effluent water, but not in the receiving (i.e., water, sediment or biota). However, the compounds were detected in sludge, suggesting particle sorption as an important mechanism for waste water removal. *Cefotaxime, irinotecan, ofloxacin, 6-OH-paclitaxel, paracetamol,* and *spiramycin* were detected in hospital and STP effluent water, but not in the receiving, and not in the sludge. This suggests that some kind of chemical or biological transformation process occur in the STP. The large difference ($>10^4$ ng/L) between detected concentrations and previously reported ecotoxicological concentrations eliminate *iodixanol, iohexol,* and *iopromide* for further consideration.

The following compounds are of *some environmental concern* as they fulfil most, but not all, criteria given above: *tamoxifen* and *morphine*. *Tamoxifen* is unique in the present study, as it has only been detected in one sample, in a mussel from Tromsøsund. However, the ecotoxicological effects of *tamoxifen* are not known. *Tamoxifen* is therefore a compound of *some environmental concern*. The ecotoxicological effects of *morphine* are not known, but it

was found in receiving water in Oslofjord. *Morphine* is therefore a compound of *some environmental concern*.

The following compounds are of *environmental concern* due to their presence in receiving waters or sediments and their reported ecotoxicological effects: *carbamazepine*, *naproxen*, and *propranolol*. They are all detected in receiving waters or sediments at concentrations higher than 1/1000 than the reported most toxic ecotoxicological effect. Therefore, it cannot be excluded that these pharmaceuticals have a negative effect on aquatic organisms.

Aquaculture medicines

Cypermethrin, deltamethrin, fenbendazole, flumequine, and *praziquantel* were not detected and will not be further discussed. The large difference ($>10^4$ ng/L) between detected concentrations and previously reported ecotoxicological concentrations eliminate *emamectin* and *oxolinic acid* for further consideration. Hence, none of the aquaculture medicines included in this screening does cause any *environmental concern*.

Personal care products

Avobenzone was not detected and will not be further discussed. *Cocoamidopropyl betaine* was detected in STP effluent water, but not in the receiving (i.e., water, sediment or biota). However, the compound was detected in sludge, suggesting particle sorption as an important mechanism for waste water removal.

The following compounds are of *some environmental concern* as they fulfil most, but not all, criteria given above: *EDTA, butyl paraben, lauryl sulfate,* and *laureth sulfate*. The primary mechanism by which *EDTA* is ecotoxicological is supposed to be its ability to facilitate the uptake of non-desired metal ions. However, as all di- and trivalent metal cations will compete for *EDTA* binding, the equilibrium constant and the different cation concentrations are crucial for the toxic effect of *EDTA*. *EDTA* is therefore only ranked as a compound of *some environmental concern*. *Butyl paraben* is detected at 2 - 4 ng/L in three out of four positive receiving water samples. In the fourth sample, a concentration of 900 ng/L is reported. *Butyl paraben* is very sensitive for lab- and sample contamination, and the divergence in concentration could be due to contamination. If the 900 ng/L sample is excluded, the difference between detected and toxic concentration is $>10^4$, but as *butyl paraben* is a weak endocrine disruptor, it is still regarded as a compound of *some environmental concern*. The detergent *sodium lauryl sulfate* may undergo β-oxidation mediated by *Pseudomonas* sp., and for *sodium laureth sulfate*, an omega-oxidation is observed. Both compounds are thus biologically degraded reducing their status to *some environmental concern*.

The following compounds are of *environmental concern* due to their presence in receiving waters or sediments and their reported ecotoxicological effects: *cetrimonium* and *diethyl phthalate*. They are both detected in receiving waters or sediments at concentrations higher than 1/1000 of the reported most toxic ecotoxicological effect. Therefore, it cannot be excluded that these PPCPs have a negative effect on aquatic organisms.

It must be emphasized that the results obtained are from a screening study, which only gives a snapshot of the reality. Hence, there is too little evidence to conclude that the compounds not detected in this screening are not present in the environment, despite sampling from locations likely to contain the non-detected compounds. A complete monitoring program would have provided more conclusive evidence.

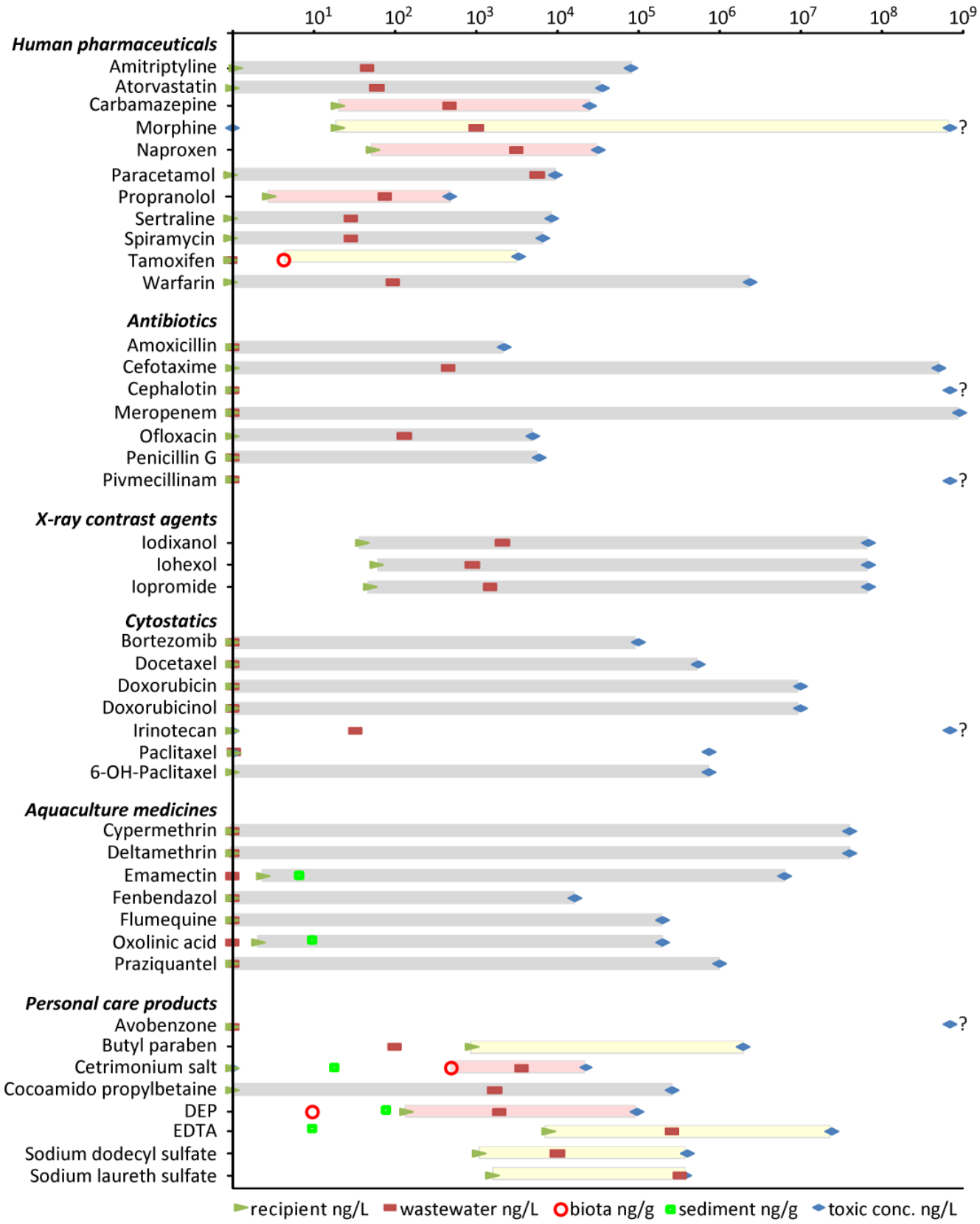

Figure 14: The figure present a summary of the highest receiving water (▶), highest waste water (▬), highest sediment (■), highest biota (O), and lowest toxic (◆) concentration for the analysed compounds. Note that the concentration axis (abscissa) is logarithmic. Yellow shading indicates a compound for which there is some environmental concern. A red shading indicates a compound of environmental concern. A grey shading indicates an negligible environmental concern for that specific compound.

No environmental concern:
General pharmaceuticals: amitriptyline, atorvastatin, paracetamol, sertraline, spiramycin, and warfarin;
Hospital-use pharmaceuticals: amoxicillin, cefotaxime, cefalotin, meropenem, ofloxacin, penicillin G, pivmecillinam, the x-ray contrasting agents iohexol, iodixanol, iopromide, and the cytostatics doxorubicin, irinotecan, bortezomib, docetaxel, paclitaxel, (and the metabolites doxorubicinol and 6-OH-paclitaxel);
Aquaculture medicines: cypermethrin, deltamethrin, emamectin, fenbendazole, flumequine, oxolinic acid, and praziquantel;
Personal care products: avobenzone and cocoamidopropyl betaine.

Some environmental concern:
General pharmaceuticals: Tamoxifen and morphine;
Personal care products: EDTA, butyl paraben, sodium dodecyl sulphate (SDS), and sodium laureth sulphate (SDSEO).

Environmental concern:
General pharmaceuticals: carbamazepine, naproxen, propranolol;
Personal care products: cetrimonium, and diethyl phthalate.

For compounds which are categorized as of some environmental concern or of environmental concern, toxic and other adverse effects on aquatic organisms and on the aquatic environment cannot be excluded. The environmental levels and effects of these compounds should therefore be studied in more detail.

Other studies indicate that the Northern environments may be more vulnerable for negative effects from the discharge of pharmaceuticals. A more thorough monitoring of the receiving waters, sediment and biota for pharmaceuticals and selected metabolites, is thus recommended.

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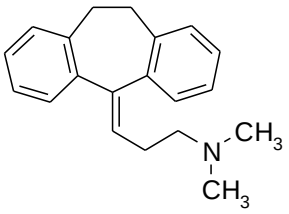
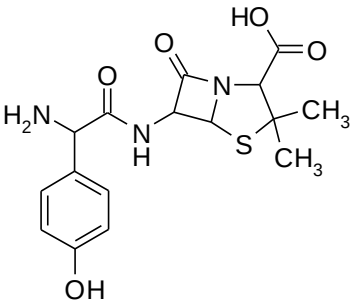
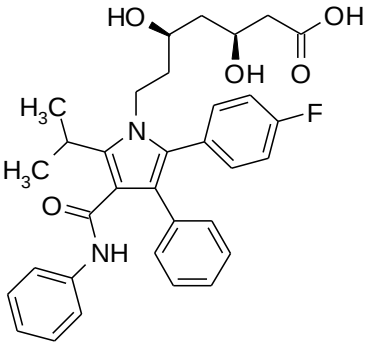
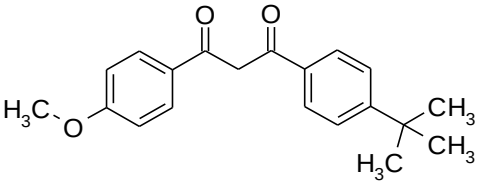
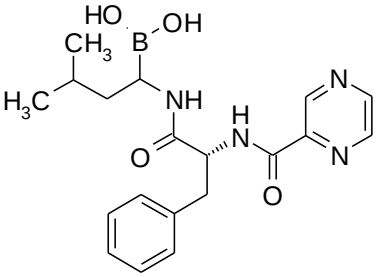
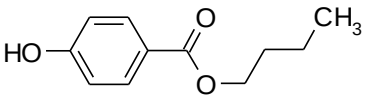
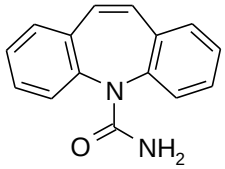
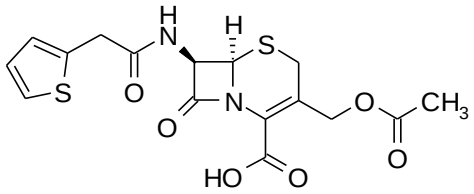
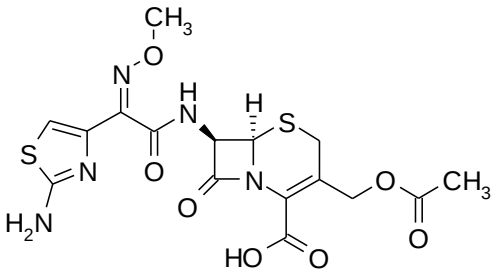
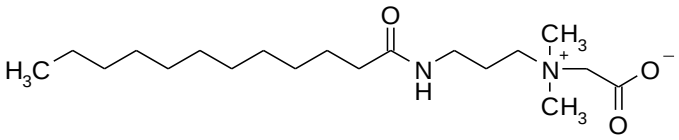
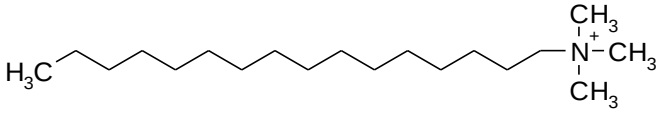
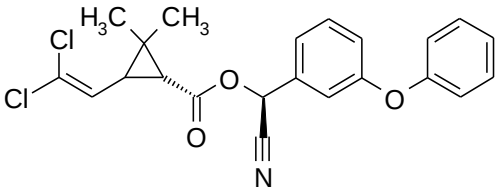
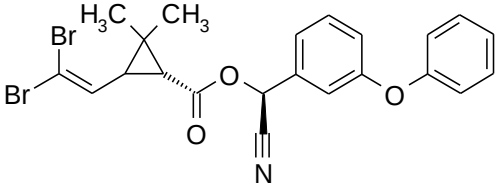
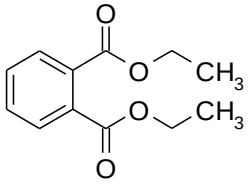
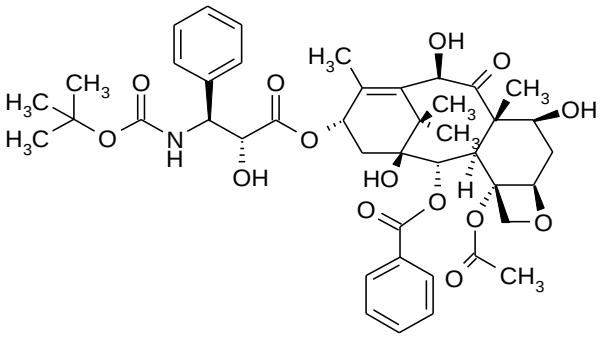
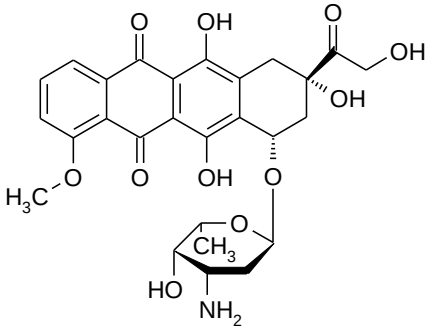
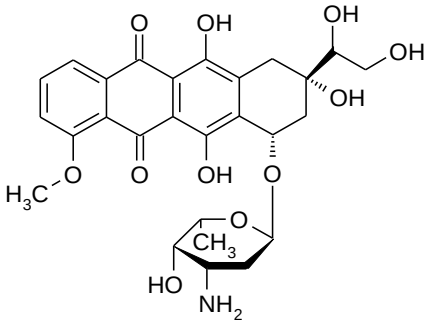
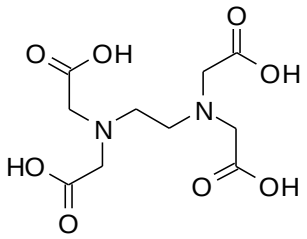
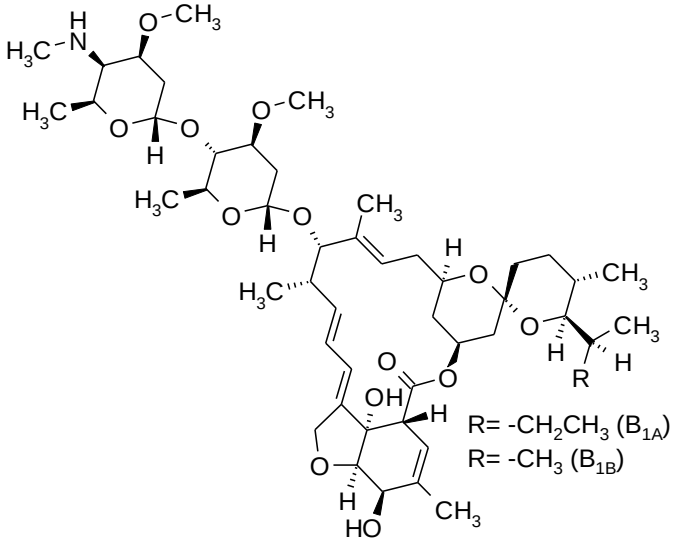
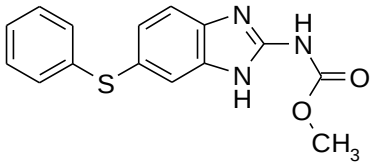
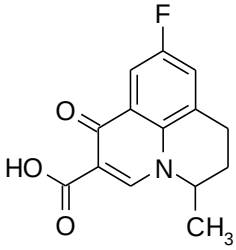
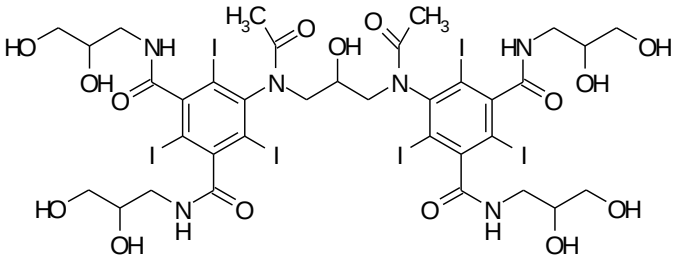
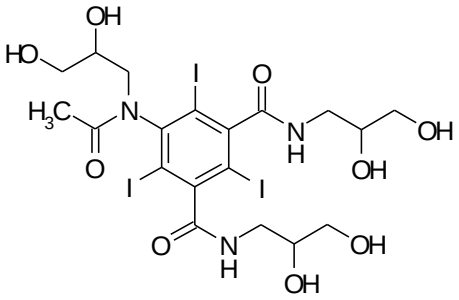
8. Appendix 1 – Chemical identity of measured compounds

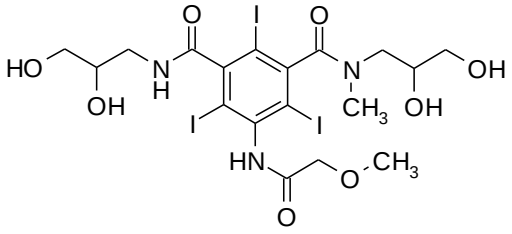
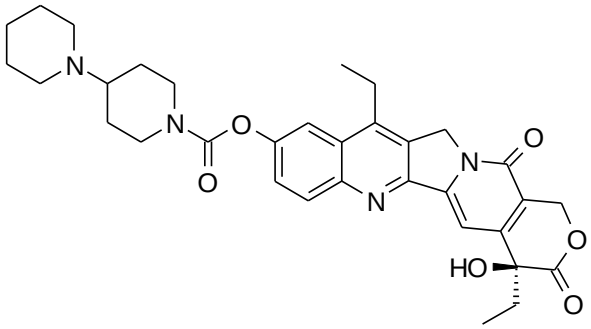
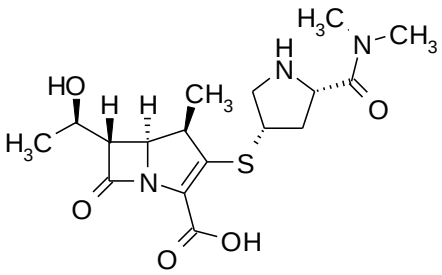
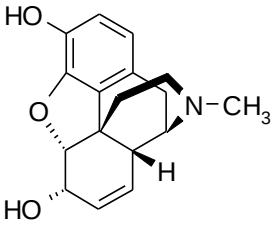
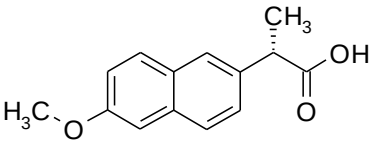
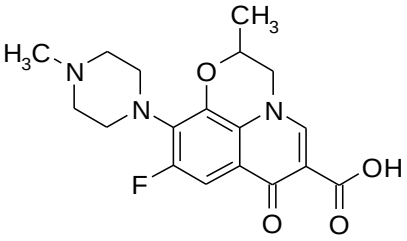
CAS numbers as the compounds are drawn here, and not as their corresponding salts, unless indicated.

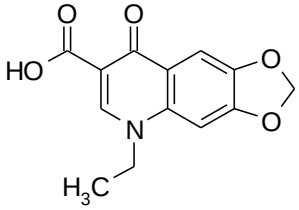
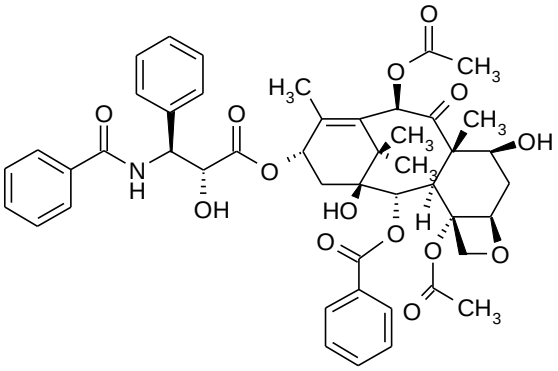
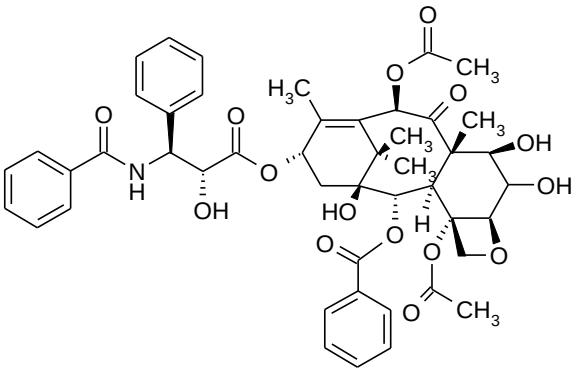
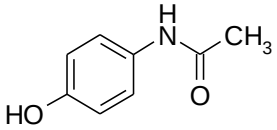
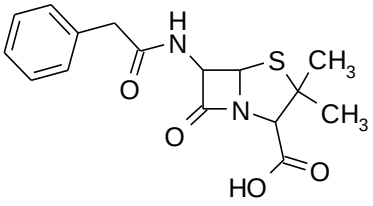
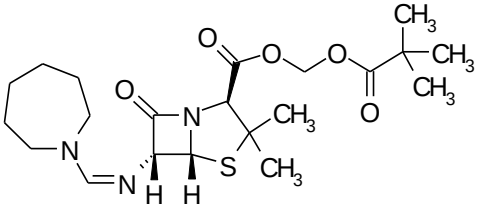
Compound	Abbreviation	CAS-no	Structure
Amitriptyline		50-48-6 N06AA09	
Amoxicillin		26787-78-0 J01CA04	
Atorvastatin		134523-00-5 C10AA05	
Avobenzone		70356-09-1	
Bortezomib		179324-69-7 L01XX32	
Butyl paraben		94-26-8	

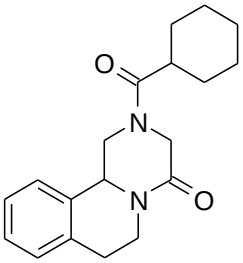
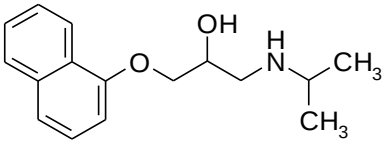
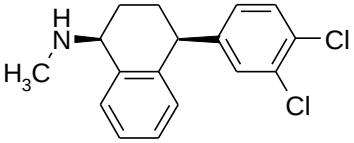
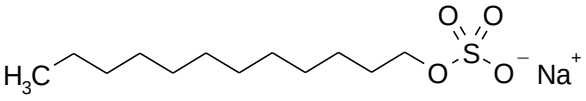
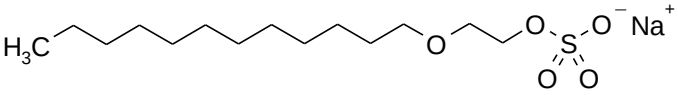
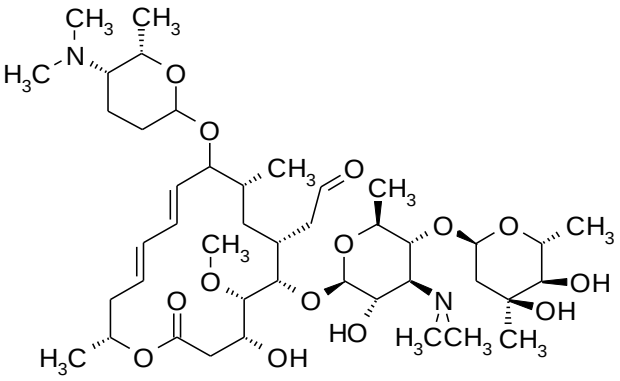
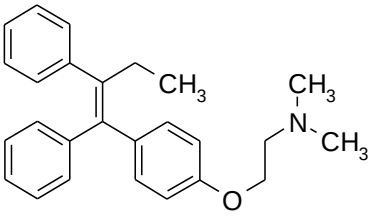
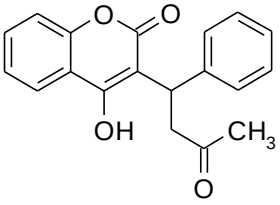
Compound	Abbreviation	CAS-no	Structure
Carbamazepine		298-46-4 N03AF01	
Cefalotin		153-61-7 J01D B03	
Cefotaxime		63527-52-6 J01DD01	
Cocoamido-propyl betaine	CAPB	4292-10-8	
Cetrimonium Br Cetrimonium Cl		57-09-0 112-02-7	
Cypermethrin		52315-07-8 QP53AC08	
Deltamethrin		52918-63-5 QP53AC11	
Diethylphthalate	DEP	84-66-2	

Compound	Abbreviation	CAS-no	Structure
Docetaxel		114977-28-5 L01CD02	
Doxorubicin		23214-92-8 L01DB01	
Doxorubicinol		54193-28-1 N/A	
EDTA	EDTA	60-00-4	

Compound	Abbreviation	CAS-no	Structure
Emamectin		119791-41-2 QP54AA06	
Fenbendazole		43210-67-9 QP52AC13	
Flumequine		42835-25-6 QJ01MB07	
Iodixanol		92339-11-2 V08AB09	
Iohexol		66108-95-0 V08AB02	

Compound	Abbreviation	CAS-no	Structure
Iopromide		73334-07-3 V08AB05	
Irinotecan		100286-90-6 L01XX19	
Meropenem		119478-56-7 J01DH02	
Morphine		57-27-2 N02AA01	
Naproxen		22204-53-1 M01AE02	
Ofloxacin		82419-36-1 J01MA01	

Compound	Abbreviation	CAS-no	Structure
Oxolinic acid		14698-29-4 QJ01MB91	
Paclitaxel		33069-62-4 L01C D01	
6-OH-Paclitaxel		153212-75-0 N/A	
Paracetamol		103-90-2 N02BE01	
Penicillin G		61-33-6 J01CE01	
Pivmecillinam		32886-97-8 J01CA08	

Compound	Abbreviation	CAS-no	Structure
Praziquantel		55268-74-1 QP54AA51	
Propranolol		525-66-6 C07AA05	
Sertraline		79617-96-2 N06AB06	
Sodium dodecyl sulfate	SDS	151-21-3	
Sodium laureth sulfate		9004-82-4	
Spiramycin		8025-81-8 J01FA02	
Tamoxifen		10540-29-1 L02BA01	
Warfarin		81-81-2 B01AA03	

9. Appendix 2 – Samples collected

Table A2a. Summary of samples; main area, sampling station and GPS coordinates, sample category and sample matrix.

Notation	Area	Station	Latitude°	Longitude°	Category	Matrix
Os_ho_1-4	Inner Oslofjord	Ullevål hospital	59°56.128	10°44.259	hospital effluent	water
Oso_tp_blank	Inner Oslofjord	VEAS WWTP	59°47.365	10°29.597	blank	water
Os_tp_eff_1-4	Inner Oslofjord	VEAS WWTP	59°47.365	10°29.597	WWTP effluent	water
Os_tp_slu_1-2	Inner Oslofjord	VEAS WWTP	59°47.365	10°29.597	WWTP effluent	sludge
Os_res_blank	Inner Oslofjord	Slemmestad bank	59°47.585	10°30.880	blank	water
Os_res_w_1-4	Inner Oslofjord	Slemmestad bank	59°47.585	10°30.880	receiving	water
Os_res_w_2	Inner Oslofjord	Slemmestad bank	59°47.531	10°30.887	receiving	water
Os_res_w_3	Inner Oslofjord	Slemmestad bank	59°47.478	10°30.895	receiving	water
Os_res_w_4	Inner Oslofjord	Slemmestad bank	59°47.424	10°30.902	receiving	water
Os_res_w_5	Inner Oslofjord	Slemmestad bank	59°47.371	10°30.909	receiving	water
Os_res_sed_1	Inner Oslofjord	Slemmestad bank	59°47.447	10°30.793	receiving	sediment
Os_res_sed_2	Inner Oslofjord	Slemmestad bank	59°47.447	10°30.793	receiving	sediment
Os_res_sed_3	Inner Oslofjord	Slemmestad bank	59°47.447	10°30.793	receiving	sediment
Os_res_bio_1	Inner Oslofjord	Gåsøya	59°51.085	10°35.341	receiving	blue mussels
Os_res_bio_2	Inner Oslofjord	Ramton	59°44.555	10°31.221	receiving	blue mussels
Tr_ho_1-4	Tromsøsund	Breivika WWTP	69°40.304	18°58.478	hospital effluent	water
Tr_tp_blank	Tromsøsund	Breivika WWTP	69°40.304	18°58.478	blank	water
Tr_tp_eff_1-4	Tromsøsund	Breivika WWTP	69°40.304	18°58.478	WWTP effluent	water
Tr_tp_sed_1-2	Tromsøsund	Breivika WWTP	69°40.304	18°58.478	WWTP effluent	sediment
Tr_res_blank	Tromsøsund	Breivika WWTP	69°40.304	18°58.478	WWTP effluent	sediment
Tr_res_w1	Tromsøsund	Tromsøy strait	69°40.452	18°59.264	blank	water
Tr_res_w1	Tromsøsund	Tromsøy strait	69°40.449	18°59.305	receiving	water
Tr_res_w2	Tromsøsund	Tromsøy strait	69°40.452	18°59.264	receiving	water
Tr_res_w3	Tromsøsund	Tromsøy strait	69°40.402	18°59.216	receiving	water
Tr_res_w4	Tromsøsund	Tromsøy strait	69°40.375	18°59.182	receiving	water
Tr_res_w5	Tromsøsund	Tromsøy strait	69°40.327	18°59.108	receiving	water
Tr_res_sed_1	Tromsøsund	Tromsøy strait	69°40.449	18°59.305	receiving	sediment
Tr_res_sed_2	Tromsøsund	Tromsøy strait	69°40.449	18°59.305	receiving	sediment
Tr_res_sed_3	Tromsøsund	Tromsøy strait	69°40.449	18°59.305	receiving	sediment
Tr_res_bio_1	Tromsøsund	Tromsøy strait	69°40.389	18°58.717	receiving	blue mussels
Tr_res_bio_2	Tromsøsund	Tromsøy strait	69°40.577	18°59.015	receiving	blue mussels
Bø_res_bl	Bømlafjord	Fish farm 1	59°36.535	05°18.995	blank	water
Bø_res_w_1	Bømlafjord	Fish farm 1	59°36.535	05°18.995	receiving	water
Bø_res_w_2	Bømlafjord	Fish farm 1	59°36.526	05°18.930	receiving	water
Bø_res_w_3	Bømlafjord	Fish farm 1	59°36.517	05°18.869	receiving	water
Bø_res_w_4	Bømlafjord	Fish farm 1	59°36.507	05°18.810	receiving	water
Bø_res_w_5	Bømlafjord	Fish farm 1	59°36.482	05°18.687	receiving	water
Bø_res_sed_1	Bømlafjord	Fish farm 1	59°36.504	05°19.019	receiving	sediment
Bø_res_sed_2	Bømlafjord	Fish farm 1	59°36.504	05°19.019	receiving	sediment
Bø_res_sed_3	Bømlafjord	Fish farm 1	59°36.504	05°19.019	receiving	sediment
Bø_res_sed_4	Bømlafjord	Fish farm 1	59°36.504	05°19.019	receiving	sediment
Bø_res_sed_5	Bømlafjord	Fish farm 1	59°36.504	05°19.019	receiving	sediment
Bø_res_bio_1	Bømlafjord	Fish farm 1	59°36.727	05°19.142	receiving	blue mussels
Bø_res_bio_2	Bømlafjord	Fish farm 1	59°36.477	05°19.216	receiving	blue mussels
Ro_res_w-1	Romsdalsfjord	Fish farm 2	62°34.607	7°08.751	receiving	water
Ro_res_w-2	Romsdalsfjord	Fish farm 2	62°34.568	7°08.765	receiving	water
Ro_res_w-3	Romsdalsfjord	Fish farm 2	62°34.540	7°08.784	receiving	water
Ro_res_w-4	Romsdalsfjord	Fish farm 2	62°34.441	7°08.824	receiving	water
Ro_res_w-5	Romsdalsfjord	Fish farm 2	62°34.345	7°08.858	receiving	water

Rol_res_sed_1	Romsdalsfjord	Fish farm 2	62°34.607	7°08.751	receiving	sediment
Ro_res_sed_2	Romsdalsfjord	Fish farm 2	62°34.607	7°08.751	receiving	sediment
Ro_res_sed_3	Romsdalsfjord	Fish farm 2	62°34.607	7°08.751	receiving	sediment
Ro_res_sed_4	Romsdalsfjord	Fish farm 2	62°34.607	7°08.751	receiving	sediment
Ro_res_sed_5	Romsdalsfjord	Fish farm 2	62°34.607	7°08.751	receiving	sediment
Ro_res_bio_1	Romsdalsfjord	Fish farm 2	62°34.610	7°08.586	receiving	blue mussels
Ro_res_bio_2	Romsdalsfjord	Fish farm 2	62°34.610	7°08.481	receiving	blue mussels

Table A2b. Overview of sample area, location, matrix, sample depth, position and date for marine samples (see also figures X to X)

Notation	Area	Station	Category	Latitude°	Longitude°	Sample type	Sampling depth (m)
Os_ho_1	Inner Oslofjord	Ullevål hospital	hospital effluent	59°56.128	10°44.259	water	ditch
Os_ho_2	Inner Oslofjord	Ullevål hospital	hospital effluent	59°56.128	10°44.259	water	ditch
Os_ho_3	Inner Oslofjord	Ullevål hospital	hospital effluent	59°56.128	10°44.259	water	ditch
Os_ho_4	Inner Oslofjord	Ullevål hospital	hospital effluent	59°56.128	10°44.259	water	ditch
Oso_tp_blank	Inner Oslofjord	VEAS STP	blank	59°47.365	10°29.597	water	air
Os_tp_eff_1	Inner Oslofjord	VEAS STP	STP effluent	59°47.365	10°29.597	water	ditch
Os_tp_eff_2	Inner Oslofjord	VEAS STP	STP effluent	59°47.365	10°29.597	water	ditch
Os_tp_eff_3	Inner Oslofjord	VEAS STP	STP effluent	59°47.365	10°29.597	water	ditch
Os_tp_eff_4	Inner Oslofjord	VEAS STP	STP effluent	59°47.365	10°29.597	water	ditch
Os_tp_slu_1	Inner Oslofjord	VEAS STP	STP effluent	59°47.365	10°29.597	sludge	ditch
Os_tp_slu_2	Inner Oslofjord	VEAS STP	STP effluent	59°47.365	10°29.597	sludge	ditch
Os_res_blank	Inner Oslofjord	Slemmestad bank	blank	59°47.585	10°30.880	water	air
Os_res_w_1	Inner Oslofjord	Slemmestad bank	receiving	59°47.585	10°30.880	water	25
Os_res_w_2	Inner Oslofjord	Slemmestad bank	receiving	59°47.531	10°30.887	water	25
Os_res_w_3	Inner Oslofjord	Slemmestad bank	receiving	59°47.478	10°30.895	water	28
Os_res_w_4	Inner Oslofjord	Slemmestad bank	receiving	59°47.424	10°30.902	water	28
Os_res_w_5	Inner Oslofjord	Slemmestad bank	receiving	59°47.371	10°30.909	water	28
Os_res_sed_1	Inner Oslofjord	Slemmestad bank	receiving	59°47.447	10°30.793	sediment	31
Os_res_sed_2	Inner Oslofjord	Slemmestad bank	receiving	59°47.447	10°30.793	sediment	31
Os_res_sed_3	Inner Oslofjord	Slemmestad bank	receiving	59°47.447	10°30.793	sediment	31
Os_res_bio_1	Inner Oslofjord	Gåsøya	receiving	59°51.085	10°35.341	blue mussels	surface
Os_res_bio_2	Inner Oslofjord	Ramton	receiving	59°44.555	10°31.221	blue mussels	surface
Tr_ho_1	Tromsøsund	Breivika STP	hospital effluent	69°40.304	18°58.478	water	surface
Tr_ho_2	Tromsøsund	Breivika STP	hospital effluent	69°40.304	18°58.478	water	surface
Tr_ho_3	Tromsøsund	Breivika STP	hospital effluent	69°40.304	18°58.478	water	surface
Tr_ho_4	Tromsøsund	Breivika STP	hospital effluent	69°40.304	18°58.478	water	surface
Tr_tp_blank	Tromsøsund	Breivika STP	STP effluent	69°40.304	18°58.478	water	surface
Tr_tp_eff_1	Tromsøsund	Breivika STP	STP effluent	69°40.304	18°58.478	water	surface

Tr_tp_eff_2	Tromsøsund	Breivika STP	STP effluent	69°40.304	18°58.478	water	surface
Tr_tp_eff_3	Tromsøsund	Breivika STP	STP effluent	69°40.304	18°58.478	water	surface
Tr_tp_eff_4	Tromsøsund	Breivika STP	STP effluent	69°40.304	18°58.478	water	surface
Tr_tp_sed_1	Tromsøsund	Breivika STP	STP effluent	69°40.304	18°58.478	sediment	surface
Tr_tp_sed_2	Tromsøsund	Breivika STP	STP effluent	69°40.304	18°58.478	sediment	surface
Tr_res_blank	Tromsøsund	Tromsø strait	blank	69°40.452	18°59.264	water	surface
Tr_res_w1	Tromsøsund	Tromsø strait	receiving	69°40.449	18°59.305	water	28
Tr_res_w1	Tromsøsund	Tromsø strait	receiving	69°40.452	18°59.264	water	28
Tr_res_w2	Tromsøsund	Tromsø strait	receiving	69°40.402	18°59.216	water	28
Tr_res_w3	Tromsøsund	Tromsø strait	receiving	69°40.375	18°59.182	water	28
Tr_res_w4	Tromsøsund	Tromsø strait	receiving	69°40.327	18°59.108	water	28
Tr_res_sed_1	Tromsøsund	Tromsø strait	receiving	69°40.449	18°59.305	sediment	30
Tr_res_sed_2	Tromsøsund	Tromsø strait	receiving	69°40.449	18°59.305	sediment	30
Tr_res_sed_3	Tromsøsund	Tromsø strait	receiving	69°40.449	18°59.305	sediment	30
Tr_res_bio_1	Tromsøsund	Tromsø strait	receiving	69°40.389	18°58.717	blue mussels	surface
Tr_res_bio_2	Tromsøsund	Tromsø strait	receiving	69°40.577	18°59.015	blue mussels	surface
Bø_res_bl	Bømlafjord	Fish farm 1	blank	59°36.535	05°18.995	water	air
Bø_res_w_1	Bømlafjord	Fish farm 1	receiving	59°36.535	05°18.995	water	10
Bø_res_w_2	Bømlafjord	Fish farm 1	receiving	59°36.526	05°18.930	water	10
Bø_res_w_3	Bømlafjord	Fish farm 1	receiving	59°36.517	05°18.869	water	10
Bø_res_w_4	Bømlafjord	Fish farm 1	receiving	59°36.507	05°18.810	water	10
Bø_res_w_5	Bømlafjord	Fish farm 1	receiving	59°36.482	05°18.687	water	10
Bø_res_sed_1	Bømlafjord	Fish farm 1	receiving	59°36.504	05°19.019	sediment	44
Bø_res_sed_2	Bømlafjord	Fish farm 1	receiving	59°36.504	05°19.019	sediment	44
Bø_res_sed_3	Bømlafjord	Fish farm 1	receiving	59°36.504	05°19.019	sediment	44
Bø_res_sed_4	Bømlafjord	Fish farm 1	receiving	59°36.504	05°19.019	sediment	44
Bø_res_sed_5	Bømlafjord	Fish farm 1	receiving	59°36.504	05°19.019	sediment	44
Bø_res_bio_1	Bømlafjord	Fish farm 1	receiving	59°36.727	05°19.142	blue mussels	surface, net cage
Bø_res_bio_2	Bømlafjord	Fish farm 1	receiving	59°36.477	05°19.216	blue mussels	surface, buoy
Ro_res_w-1	Romsdalsfjord	Fish farm 2	receiving	62°34.607	7°08.751	water	10
Ro_res_w-2	Romsdalsfjord	Fish farm 2	receiving	62°34.568	7°08.765	water	10
Ro_res_w-3	Romsdalsfjord	Fish farm 2	receiving	62°34.540	7°08.784	water	10
Ro_res_w-4	Romsdalsfjord	Fish farm 2	receiving	62°34.441	7°08.824	water	10
Ro_res_w-5	Romsdalsfjord	Fish farm 2	receiving	62°34.345	7°08.858	water	10
Rol_res_sed_1	Romsdalsfjord	Fish farm 2	receiving	62°34.607	7°08.751	sediment	30
Ro_res_sed_2	Romsdalsfjord	Fish farm 2	receiving	62°34.607	7°08.751	sediment	30
Ro_res_sed_3	Romsdalsfjord	Fish farm 2	receiving	62°34.607	7°08.751	sediment	30
Ro_res_sed_4	Romsdalsfjord	Fish farm 2	receiving	62°34.607	7°08.751	sediment	30
Ro_res_sed_5	Romsdalsfjord	Fish farm 2	receiving	62°34.607	7°08.751	sediment	30
Ro_res_bio_1	Romsdalsfjord	Fish farm 2	receiving	62°34.610	7°08.586	blue mussels	surface
Ro_res_bio_2	Romsdalsfjord	Fish farm 2	receiving	62°34.610	7°08.481	blue mussels	surface

Area	Station	Sample type	Sampling depth (m)	Position	Position	Sample date
Receiving waters						
Oslofjord	VEAS WTP	Water + blank 1	25	59°47.585	10°30.880	25.08.2008 + NILU-2 28.08.2008
Oslofjord	VEAS WTP	Water 2	25	59°47.531	10°30.887	25.08.2008
Oslofjord	VEAS WTP	Water 3	28	59°47.478	10°30.895	25.08.2008
Oslofjord	VEAS WTP	Water 4	28	59°47.424	10°30.902	25.08.2008
Oslofjord	VEAS WTP	Water 5	28	59°47.371	10°30.909	25.08.2008
Oslofjord	VEAS WTP	Sediment, grab 1	31	59°47.447	10°30.793	14.08.2008
Oslofjord	VEAS WTP	Sediment, grab 2	31	59°47.447	10°30.793	14.08.2008
Oslofjord	VEAS WTP	Sediment, grab 3	31	59°47.447	10°30.793	14.08.2008
Oslofjord	Gåsøya	Blue mussels 1	surface	59°51.085	10°35.341	17.06.2008
Oslofjord	Ramton	Blue mussels 2	surface	59°44.555	10°31.221	17.06.2008
Tromsøsund	Breivika WTP	Water 1	28	69°40.449	18°59.305	23.09.2008
Tromsøsund	Breivika WTP	Water + blank 2	28	69°40.452	18°59.264	23.09.2008
Tromsøsund	Breivika WTP	Water 3	28	69°40.402	18°59.216	25.09.2008
Tromsøsund	Breivika WTP	Water 4	28	69°40.375	18°59.182	25.09.2008
Tromsøsund	Breivika WTP	Water 5	28	69°40.327	18°59.108	25.09.2008
Tromsøsund	Breivika WTP	Sediment, grab 1	30	69°40.449	18°59.305	10.11.2008
Tromsøsund	Breivika WTP	Sediment, grab 2	30	69°40.449	18°59.305	10.11.2008
Tromsøsund	Breivika WTP	Sediment, grab 3	30	69°40.449	18°59.305	10.11.2008
Tromsøsund	Breivika WTP	Blue mussels 1	surface	69°40.389	18°58.717	10.09.2008
Tromsøsund	Breivika WTP	Blue mussels 2	surface	69°40.577	18°59.015	10.09.2008
Fish farms						
Bømlafjord	Fish farm 1	Water + blank 1	10	59°36.535	05°18.995	08.09.2008
Bømlafjord	Fish farm 1	Water 2	10	59°36.526	05°18.930	08.09.2008
Bømlafjord	Fish farm 1	Water 3	10	59°36.517	05°18.869	08.09.2008
Bømlafjord	Fish farm 1	Water 4	10	59°36.507	05°18.810	08.09.2008
Bømlafjord	Fish farm 1	Water 5	10	59°36.482	05°18.687	08.09.2008
Bømlafjord	Fish farm 1	Sediment, grab 1	44	59°36.504	05°19.019	08.09.2008
Bømlafjord	Fish farm 1	Sediment, grab 2	44	59°36.504	05°19.019	08.09.2008
Bømlafjord	Fish farm 1	Sediment, grab 3	44	59°36.504	05°19.019	08.09.2008
Bømlafjord	Fish farm 1	Sediment, grab 4	44	59°36.504	05°19.019	08.09.2008
Bømlafjord	Fish farm 1	Sediment, grab 5	44	59°36.504	05°19.019	08.09.2008
Bømlafjord	Fish farm 1	Blue mussels 1	surface, net cage	59°36.727	05°19.142	08.09.2008
Bømlafjord	Fish farm 1	Blue mussels 2	surface, buoy	59°36.477	05°19.216	08.09.2008

Romsdalsfjord	Fish farm 2	Water 1	10	62°34.607	7°08.751	18.09.2008
Romsdalsfjord	Fish farm 2	Water 2	10	62°34.568	7°08.765	18.09.2008
Romsdalsfjord	Fish farm 2	Water 3	10	62°34.540	7°08.784	18.09.2008
Romsdalsfjord	Fish farm 2	Water 4	10	62°34.441	7°08.824	18.09.2008
Romsdalsfjord	Fish farm 2	Water 5	10	62°34.345	7°08.858	18.09.2008
Romsdalsfjord	Fish farm 2	Sediment, grab 1	30	62°34.607	7°08.751	18.09.2008
Romsdalsfjord	Fish farm 2	Sediment, grab 2	30	62°34.607	7°08.751	18.09.2008
Romsdalsfjord	Fish farm 2	Sediment, grab 3	30	62°34.607	7°08.751	18.09.2008
Romsdalsfjord	Fish farm 2	Sediment, grab 4	30	62°34.607	7°08.751	18.09.2008
Romsdalsfjord	Fish farm 2	Sediment, grab 5	30	62°34.607	7°08.751	18.09.2008
Romsdalsfjord	Fish farm 2	Blue mussels 1	surface	62°34.610	7°08.586	18.09.2008
Romsdalsfjord	Fish farm 2	Blue mussels 2	surface	62°34.610	7°08.481	18.09.2008

10. Appendix 3 – Measured concentrations of all samples

	Paracetamol	Naproxen	Propranolol	Carbamazepine	Amitriptyline	Spiramycin	Morphine	Sertraline	Warfarin	Tamoxifen	Atorvastatin
ng/L											
Off VEAS blank	<1	2.7	<1	<2	<1	<3	<4	<2	<3	<1	<1
Off VEAS 0 m	<1	53.5	2.0	19.6	<1	<3	9.2	<2	<5	<1	<2
Off VEAS 100 m	<1	24.2	1.9	10.0	<1	<3	4.7	<2	<5	<1	<2
Off VEAS 200 m	<1	45.8	2.3	18.1	<1	<3	21.7	<2	<5	<1	<2
Off VEAS 300 m	<1	39.9	3.0	14.3	1.1	<3	18.9	<2	<5	<1	<2
Off VEAS 400 m	<1	32.4	1.6	18.2	<1	<3	14.4	<2	<5	<1	<2
Off Breivika blank	<1	3.5	<1	<1	<1	<3	<4	<2	<3	<1	<1
Off Breivika 0 m	<1	4.9	0.5	<1	<1	<3	<4	<2	<5	<1	<2
Off Breivika 50 m	<1	11.0	<1	<1	<1	<3	<4	<2	<5	<1	<2
Off Breivika 100 m	<1	5.9	0.8	<1	<1	<3	<4	<2	<5	<1	<2
Off Breivika 150 m	<1	10.8	<1	1.0	<1	<3	<4	<2	<5	<1	<2
Off Breivika 300 m	<1	9.6	1.2	<1	<1	<3	<4	<2	<5	<1	<2
Ullevål 0309-0409											
Ullevål 0809-0909											
Ullevål 0909-1009											
Ullevål 1109-1209											
Ullevål 1709-1809											
VEAS effluent blank	<1	<1	<1	<1	<1	<3	<15	<2	<3	<1	<1
VEAS effluent 0209-0309	887	1091	22.5	413	20.9	30.4	980	7.1	10.8	<1	56.1
VEAS effluent 0309-0409	189	794	42.4	449	18.3	30.4	529	11.5	36.6	<1	47.6
VEAS effluent 1109-1209	680	1054	26.8	474	24.5	20.3	784	3.9	31.0	<1	0.0
VEAS effluent 1509-1609	<5	62.3	23.2	236	22.9	9.0	<20	10.7	70.6	<1	45.6
UNN effluent 2508-2608											
UNN effluent 2608-2708											
UNN effluent 2708-2808											
UNN effluent 2808-2908											
Breivika effl. blank 25-2608	<1	<1	1.6	<1	<1	<3	<4	<2	<3	<1	<1
Breivika effluent 2508-2608	6013	3157	77.7	252	43.2	0.0	864	0.0	104	<1	<2
Breivika effluent 2608-2708	4900	1661	62.4	256	34.9	0.0	547	30.5	36.3	<1	<2
Breivika effluent 2708-2808	4996	1569	54.9	395	46.5	0.0	313	19.4	54.5	<1	23.2
Breivika effluent 2808-2908	3294	1192	47.9	274	29.6	0.0	216	5.4	48.2	<1	<2
ng/g (d/w)											
VEAS sediment 1 140808	<4	<5	<5	<1	<1	<3	<6	<4	<10	<2	<5
VEAS sediment 2 140808	<4	<5	<5	<1	<1	<3	<6	<3	<5	<4	<5
VEAS sediment 3 140808	<4	<5	<5	<1	<1	<3	<6	<3	<10	<1	<5
Breivika sediment 1 100908	<2	<5	<1	<1	<1	<3	<10	<1.5	<10	<1	<5
Breivika sediment 2 100908	<2	<5	<1	<1	<1	<2	<10	<1.5	<10	<1	<5
Breivika sediment 3 100908	<2	<5	<1	<1	<1	<2	<10	<1.5	<10	<1	<5
VEAS sludge 040408	<6	11.4	22.9	86.4	17.2	<7	<9	33.0	17.0	2.1	9.9
VEAS sludge 120908	<6	10.8	30.3	101	29.0	<7	<9	43.7	<10	<1	8.4
Breivika sludge 1	<3	8.1	12.9	117	15.9	<4	<8	13.3	10.2	<1	<5
Breivika sludge 2	<3	17.0	12.3	196	12.7	<4	<8	12.7	15.1	0.9	<5
ng/g (w/w)											
Oslo - Ramton mussel	<15	<5	<5	<5	<5	<5	<10	<5	<25	<5	<5
Oslo - Gåsøya mussel	<15	<5	<5	<5	<5	<5	<18	<5	<17	<10	<5
Tomsø Mussel 1	<15	<5	<5	<5	<5	<5	<15	<5	<20	5,0	<5
Tomsø Mussel 2											

	Amoxicillin	Cefotaxime	Cephalotin	Meropenem	Ofloxacin	Penicillin G	Pivmecillinam	Iohexol	Iodixanol	Iopromide
ng/L										
Off VEAS blank	<29	<0.66	<2.9	<6.2	<0.76	<2.6	<0.15	<20	<20	<2
Off VEAS 0 m	<14	<0.85	<4.2	<4.3	<1.1	<4.0	<0.25	58	38	6
Off VEAS 100 m	<19	<1.2	<5.0	<4.7	<1.7	<5.5	<0.31	18	12	<2
Off VEAS 200 m	<24	<1.0	<4.4	<6.4	<1.3	<3.7	<0.28	13	<20	3
Off VEAS 300 m	<106	<1.6	<6.6	<30	<2.1	<6.1	<0.45	10	13	9
Off VEAS 400 m	<53	<1.0	<4.2	<18	<1.3	<4.2	<0.29	<20	<20	3
Off Breivika blank										
Off Breivika 0 m	<21	<0.99	<3.4	<6.4	<1.03	<2.8	<0.21	<20	<20	30
Off Breivika 50 m	<20	<1.3	<4.4	<6.4	<1.34	<4.2	<0.27	<20	<20	26
Off Breivika 100 m	<14	<1.9	<7.3	<2.9	<1.89	<7.1	<0.41	<20	<20	13
Off Breivika 150 m	<23	<1.0	<4.3	<5.6	<1.11	<4.0	<0.24	<20	<20	54
Off Breivika 300 m	<38	<1.2	<4.8	<9.3	<1.20	<3.5	<0.24	<20	14	14
Ullevål 0309-0409	<30	33	<9.2	<9.0	<4.5	<63	<1.3	330	<20	150
Ullevål 0809-0909	<91	441	<33	<32	<19	<65	<3.5	196	16	13
Ullevål 0909-1009	<19	85	<7.4	<7.0	129	<49	<0.83	128	<20	<4
Ullevål 1109-1209	<206	244	<77	<71	<48	<204	<11	117	<20	72
Ullevål 1709-1809	<21	32	<7.3	<7.3	<5.0	<36	<1.1			
VEAS effluent blank	<16	<1.6	<4.3	<2.3	<1.3	<6.1	<0.34	<20	<20	<2
VEAS effluent 0209-0309	<7.8	42	<3.0	<2.3	<1.6	<19	<0.47	283	204	24
VEAS effluent 0309-0409	<7.0	36	<2.3	<2.5	<1.2	<14	<0.29	309	198	13
VEAS effluent 1109-1209	<8.7	42	<2.6	<2.6	<1.5	<18	<0.38	252	149	21
VEAS effluent 1509-1609	<17	53	<4.5	<4.5	<2.1	<15	<0.62	216	103	7
UNN effluent 2508-2608	<118	325	<126	<85	<38	<86	<7.7	887	1537	1298
UNN effluent 2608-2708	<81	<15	<83	<55	<24	<135	<4.2	602	1200	1523
UNN effluent 2708-2808	<201	62	<212	<104	<49	<110	<11	248	1898	1245
UNN effluent 2808-2908	<120	75	<106	<74	<25	<107	<6.5	693	2103	1219
Breivika effl. blank 25-2608	<25	<4.6	<20	<7.2	<4.4	<23	<0.85	<20	<20	<2
Breivika effluent 2508-2608	<173	113	<155	<100	<42	<100	<9.6	920	1498	1293
Breivika effluent 2608-2708	<138	577	<119	<94	<46	<112	<11	890	1652	1357
Breivika effluent 2708-2808	<76	334	<77	<37	<21	<66	<4.0	339	1650	1158
Breivika effluent 2808-2908	<122	401	<111	<72	<39	<84	<8.3	575	1732	957
ng/g (d/w)										
VEAS sediment 1 140808	<3.2	<0.53	<2.0	<1.2	<0.51	<1.6	<0.21	13	<1	2
VEAS sediment 2 140808	<5.2	<0.74	<2.9	<1.7	<0.73	<2.3	<0.32	<0.8	<1	<0.5
VEAS sediment 3 140808	<4.5	<0.81	<2.6	<2.2	<0.83	<2.5	<0.29	<0.8	7	1
Breivika sediment 1 100908	<1.7	<0.28	<0.87	<0.62	<0.21	<0.71	<0.09	<0.8	7	1
Breivika sediment 2 100908	<2.1	<0.33	<1.0	<0.95	<0.27	<0.89	<0.10	<0.8	5	1
Breivika sediment 3 100908	<1.7	<0.19	<0.87	<0.75	<0.18	<0.51	<0.06	<0.8	<1	<0.5
VEAS sludge 040408	<18	<3.6	<9.1	<8.2	<6.9	<8.4	<1.8	<0.8	<1	<0.5
VEAS sludge 120908	<35	<4.8	<11	<13	<6.3	<9.9	<2.3	<0.8	10	<0.5
Breivika sludge 1	<145	<5.4	<14	<60	<7.0	<12	<2.3	<0.8	<1	<0.5
Breivika sludge 2	<232	<5.3	<12	<88	<6.1	<12	<1.2	<0.8	<1	<0.5
ng/g (w/w)										
Oslo - Ramton mussel	<19	<1.4	<4.4	<10	<1.6	<5.3	<0.70			
Oslo - Gåsøya mussel	<16	<1.2	<3.9	<6.5	<1.4	<5.0	<0.55			
Tomsø Mussel 1	<11	<0.48	<1.7	<4.6	<0.59	<2.1	<0.26			
Tromsø Mussel 2	<17	<0.61	<2.5	<9.4	<0.63	<2.0	<0.38			

	Doxorubicin	Doxorubicinol	Irinotecan	Bortezomib	Docetaxel	Paclitaxel	6-OH-Paclitaxel
Off VEAS blank	<0.7	<1.1	<0.1	<14	<2.2	<0.9	<1.6
Off VEAS 0 m	<1.0	<1.1	<0.1	<14	<1.0	<1.0	<1.2
Off VEAS 100 m	<0.7	<1.5	<0.1	<13	<1.7	<0.7	<1.6
Off VEAS 200 m	<1.3	<1.8	<0.2	<12	<1.5	<0.6	<1.8
Off VEAS 300 m	<1.5	<3.7	<0.1	<13	<0.8	<1.9	<1.7
Off VEAS 400 m	<0.9	<2.1	<0.2	<11	<1.1	<1.1	<1.2
Off Breivika blank	<0.8	<1.3	<0.1	<6.4	<1.1	<0.5	<1.2
Off Breivika 0 m	<1.0	<1.3	<0.1	<6.0	<0.9	<0.9	<1.8
Off Breivika 50 m	<1.2	<2.2	<0.2	<11	<2.6	<1.9	<4.9
Off Breivika 100 m	<0.7	<1.4	<0.1	<5.8	<2.0	<1.1	<2.3
Off Breivika 150 m	<0.6	<1.4	<0.1	<5.3	<0.5	<0.5	<1.4
Off Breivika 300 m	<8.2	<9.3	<0.9	<137	<32	<3.9	<6.3
Ullevål 0309-0409	<8.6	<23	<0.7	<217	<28	<3.9	<8.3
Ullevål 0809-0909	<8.4	<15	13	<89	<16	<3.5	<6.4
Ullevål 0909-1009	<8.5	<12	30	<118	<15	<3.5	<6.1
Ullevål 1109-1209	<7.3	<11	35	<373	<8.1	<5.9	<14
Ullevål 1709-1809	<8.8	<20	<1.2	<248	<14	<6.2	<9.7
VEAS effluent blank	<6.7	<11	<0.8	<215	<4.9	<5.4	<13
VEAS effluent 0209-0309	<6.3	<10	<0.7	<275	<7.8	<4.7	<14
VEAS effluent 0309-0409	<3.1	<5.7	<0.8	<20	<2.0	<1.4	<2.9
VEAS effluent 1109-1209	<7.3	<12	<0.7	<378	<3.9	<3.9	<6.1
VEAS effluent 1509-1609	<6.1	<10	<0.6	<311	<3.2	<2.0	<3.7
UNN effluent 2508-2608	<6.1	<42	<0.7	<513	<3.4	<3.6	<5.3
UNN effluent 2608-2708	<5.9	<11	<1.2	<512	<7.0	<3.3	<8.0
UNN effluent 2708-2808	<3.5	<7.1	<0.7	<19	<3.5	<2.5	<4.0
UNN effluent 2808-2908	<8.7	<11	<0.9	<244	<6.3	<4.9	<8.2
Breivika effl. blank 25-2608	<6.7	<11	<0.8	<244	<7.3	<3.8	<8.0
Breivika effluent 2508-2608	<8.7	<15	<1.0	<234	<4.6	<5.8	<6.6
Breivika effluent 2608-2708	<8.1	<14	14	<201	<5.9	<5.4	<5.1
Breivika effluent 2708-2808	<0.7	<1.1	29	<14	<2.2	<0.9	35
Breivika effluent 2808-2908	<1.0	<1.1	<0.1	<14	<1.0	<1.0	38
ng/g (d/w)							
VEAS sediment 1 140808	<85	<200	<750	<25	<40	<20	<45
VEAS sediment 2 140808	<85	<200	<750	<25	<40	<20	<45
VEAS sediment 3 140808	<85	<200	<750	<25	<40	<20	<45
Breivika sediment 1 100908	<85	<200	<750	<25	<40	<20	<45
Breivika sediment 2 100908	<85	<200	<750	<25	<40	<20	<45
Breivika sediment 3 100908	<85	<200	<750	<25	<40	<20	<45
VEAS sludge 040408	2607	<3500	<1100	<1200	<500	579	<650
VEAS sludge 120908	5571	<3500	<1100	<1200	<500	640	<650
Breivika sludge 1	1450	<3500	<1100	<1200	<500	<300	<650
Breivika sludge 2	<1400	<3500	<1100	<1200	<500	<300	<650
ng/g (w/w)							
Oslo - Ramton mussel							
Oslo - Gåsøya mussel							
Tomsø Mussel 1							
Tromsø Mussel 2							

	DEP	Butyl paraben	Avobenzone	EDTA	Sodium dodecyl sulfate	Sodium laureth sulfate	Cocoamido propylbetaine	Cetrimonium salt
Off VEAS blank								
Off VEAS 0 m	21,609	<2	<2	7901,6	55	110	<10	<40
Off VEAS 100 m	18,839	2,2777	<2	3715,2	61	110	<10	<40
Off VEAS 200 m	14,022	3,4807	<2	7559,0	<40	<50	<10	<40
Off VEAS 300 m								
Off VEAS 400 m								
Off Breivika blank								
Off Breivika 0 m	17,199	<2	<2	96,47	520	210	<10	<40
Off Breivika 50 m	12,358	2,9211	<2	215,27	<40	<40	<10	<40
Off Breivika 100 m	138,90	912,27	<2	6039,0	1100	1600	<10	<40
Off Breivika 150 m								
Off Breivika 300 m								
Ullevål 0309-0409								
Ullevål 0809-0909								
Ullevål 0909-1009								
Ullevål 1109-1209								
Ullevål 1709-1809								
VEAS effluent blank								
VEAS effluent 0209-0309	<10	<2	<2	310000	<300	<600	<50	<40
VEAS effluent 0309-0409	20,712	<2	<2	240000	<300	<600	<50	<40
VEAS effluent 1109-1209	<10	<2	<2	260000	<300	<600	<50	<40
VEAS effluent 1509-1609								
UNN effluent 2508-2608								
UNN effluent 2608-2708								
UNN effluent 2708-2808								
UNN effluent 2808-2908								
Breivika effl. blank 25-2608								
Breivika effluent 2508-2608	1774,6	76,657	<2	120000	9800	320000	<200	3500
Breivika effluent 2608-2708	1809,7	74,204	<2	79387	10000	300000	<200	3100
Breivika effluent 2708-2808	1935,1	97,166	<2	130000	9600	230000	<200	3600
Breivika effluent 2808-2908								
ng/g (d/w)								
VEAS sediment 1 140808	< 20	< 4	<5	<10	<40	<80	<20	17
VEAS sediment 2 140808	87	< 4	<5	10	<40	<80	<20	8,1
VEAS sediment 3 140808	< 20	< 4	<5	<10	<40	<80	<20	14
Breivika sediment 1 100908	< 20	< 4	<5	<10	<40	<80	<20	<4
Breivika sediment 2 100908	< 20	< 4	<5	<10	<40	<80	<20	<4
Breivika sediment 3 100908	< 20	< 4	<5	<10	<40	<80	<20	<4
VEAS sludge 040408	< 20	< 4	<20	1100	490	520	73	12000
VEAS sludge 120908	< 50	< 4	<20	600	350	370	72	15000
Breivika sludge 1	89	< 4	<20	280	3400	60000	1700	3300
Breivika sludge 2	51	< 4	<20	390	3200	58000	1500	3600
ng/g (w/w)								
Oslo - Ramton mussel	9,3	<4	<5	<15				<5
Oslo - Gåsøya mussel	<4	<4	<5	<15				9,5
Tomsø Mussel 1	<4	<4	<5	<15				400
Tromsø Mussel 2	<4	<4	<5	<15				500

	Oxolinic acid	Flumequine	Fenbendazol	Praziquantel	Emamectin	Cypermethrin	Deltamethrin
Fish farm 1 1 080908	<1	<1	<2	<3	<1	<2	<10
Fish farm 1 2 080908	1,2	<1	<2	<3	<1	<2	<10
Fish farm 1 3 080908	<1	<1	<2	<3	<1	<2	<10
Fish farm 1 4 080908	<1	<1	<2	<3	<1	<2	<10
Fish farm 1 5 080908	1,1	<1	<2	<3	<1	<2	<10
Fish farm 2 1 180908 anlegg	1,7	<1	<2	<3	<1	<2	<10
Fish farm 2 2 180908 50 m	1,8	<1	<2	<3	<1	<2	<10
Fish farm 2 3 180908 100 m	2,1	<1	<2	<3	<1	<2	<10
Fish farm 2 4 180908 300 m	1,5	<1	<2	<3	<1	<2	<10
Fish farm 2 5 180908 500 m	<1	<1	<2	<3	<1	<2	<10
ng/g (d/w)							
Fish farm 1 1 sediment 080908	1,3	<1	<3	<3	2,4	<5	<15
Fish farm 1 2 sediment 080908	1,2	<1	<3	<3	<2	<5	<15
Fish farm 1 3 sediment 080908	<1	<1	<3	<3	<2	<5	<15
Fish farm 1 4 sediment 080908	<1	<1	<3	<3	2,3	<5	<15
Fish farm 1 5 sediment 080908	1,2	<1	<3	<3	<2	<5	<15
Fish farm 2 1 sediment 180908	4,7	<1	<3	<3	2,1	<5	<15
Fish farm 2 2 sediment 180908	4,7	<1	<3	<3	<2	<5	<15
Fish farm 2 3 sediment 180908	5,7	<1	<3	<3	5,7	<5	<15
Fish farm 2 4 sediment 180908	11,2	<1	<3	<3	<2	<5	<15
Fish farm 2 5 sediment 180908	10,3	<1	<3	<3	6,5	<5	<15
ng/g (w/w)							
Fish farm 1 Mussel 1	<2	<1	<3	<3	<2	<5	<15
Fish farm 1 Mussel 2	<2	<1	<3	<3	<2	<5	<15
Fish farm 2 Mussel 1	<2	<1	<3	<3	<2	<5	<15
Fish farm 2 Mussel 2	<2	<1	<3	<3	<2	<5	<15

Statlig program for forurensningsovervåking

Statens forurensningstilsyn (SFT)
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Oppdragstakers prosjektansvarlig	Kontaktperson SFT	TA-nummer
Martin Schlabach	Bård Nordbø	2508/2009

	År	Sidetall	SFTs kontraktnummer
	2009	114	5009040

Utgiver	Prosjektet er finansiert av
NILU	SFT

Forfatter(e)
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Tittel - norsk og engelsk
Environmental screening of selected organic compounds 2008
Human and hospital-use pharmaceuticals, aquaculture medicines and personal care products.

Kartlegging av utvalgte stoffer i legemidler, kosmetikk og veterinærlegemidler brukt i akvakultur,
Screening 2008

Sammendrag – summary
On behalf of SFT, NILU, NIVA, and IVL monitored pharmaceuticals, hospital-use pharmaceuticals, aquaculture medicines and personal care products in samples from hospital effluent water, wastewater treatment facilities, seawater, marine sediment, and blue mussels in samples collected in 2008 as a part of a screening. The detected concentrations of the compounds included in this report were compared with their known ecotoxicological effect concentrations. For most compounds toxicity data are available for a few species, therefore a safety factor of 1000 was used. Based on this simple risk assessment, the compounds tamoxifen, morphine, EDTA, butyl paraben, lauryl sulfate, and laureth sulfate are of some environmental concern. The compounds carbamazepine, cetrimonium, diethyl phthalate, naproxen, and propranolol are of environmental concern due to their presence in receiving compartments and their reported ecotoxicological effects.

På vegne av SFT har NILU, NIVA og IVL monitorert legemidler, sykehusfarmasøytika, veterinærmedisiner of personlig pleieprodukter i prøver fra avløpsvann fra sykehus og kloakkrenseanlegg, slam, sjøvann, marine sedimenter og blåskjell. Prøvene ble hentet i 2008 i et screeningprosjekt.De påviste konsentrasjonene av forbindelsene som er omfattet av denne rapporten ble sammenlignet med deres kjente økotoksiske konsentrasjoner. For de fleste forbindelser er kun toksisiteten for et par arter kjent, og derfor ble en sikkerhetsfaktor på 1000 inkludert i risikoanalysen. Basert på dette ble tilstedeværelsen av forbindelsene tamoksifen, morfin, EDTA, butyl paraben, laurylsulfat og lauretsulfat vurdert å være av en viss miljømessig bekymring. Forbindelsene karbamazepin, cetrimonium, dietylftalat, naproksen og propranolol er alle betenkelige med tanke på konsentrasjonene som er detektert og sammenlignet med deres kjente økotoksiske effekter.

4 emneord	4 subject words
Legemidler, narkotika, kosmetiske produkter, miljø	Pharmaceuticals, narcotics, personal care products, environment

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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

Overvåkingsprogrammmet 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. SFT er ansvarlig for gjennomføringen av overvåkningsprogrammet.

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