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Palette of measures on cost-effective management options to reduce discharges, emissions, and losses of hazardous substances

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Baltic Marine Environment Protection Commission

DRAFT PALETTE OF MEASURES ON COST-EFFECTIVE MANAGEMENT OPTIONS TO REDUCE DISCHARGES, EMISSIONS AND LOSSES OF HAZARDOUS SUBSTANCES

The document is based on the final outcomes of the Project on Control of hazardous substances in the Baltic Sea region (COHIBA).
It is intended to provide information on the Hazardous Substances component of the HELCOM Baltic Sea Action Plan. The Palette contains both source-reduction and end-of-pipe solutions, represented by technical, managerial and legislative measures.

Disclaimer

This final palette reflects the state of discussions at expert level at the time of its drafting in COHIBA project. The technical document is of a non-binding nature and aims at facilitating co-operation between Member States and EU that are parties to the Helsinki Convention. The document does not necessarily represent the official, formal position of any of the Member States or EU. It is not meant to prejudice the ongoing decision making process in Contracting Parties and their final conclusions later especially the implementation of Directive 2013/39/EU of the European Parliament and of the Council of 12 August 2013 amending Directives 2000/60/EC and 2008/105/EC as regards priority substances in the field of water policy.

The Palette consists of three parts:

A. Substance-specific conclusions on management measures
B. Comparison of reduction measures
C. Detailed substance-specific assessment of measures
(see references to specific substance-wise sections below)

A. Substance-specific conclusions on management measures

1. Dioxins, furans and dioxin like polychlorinated biphenols (PCDD/F)

Air deposition is the main source of PCDD/F in the Baltic Sea catchment area. Further important sources are secondary sources, such as resuspension and long range transport.

Concerning the residential sector, reduction of PCDD/F air emissions through replacement and retrofitting of household furnaces is the most important measure. This includes installation of thermic catalysers, insulation of chimneys, chimney sweeps, use of better fuel and installation of accumulators. This action can be done in a very short time for boilers with a long future service life. In addition to that, Electrostatic Precipitators (ESP) and retrofitting of old type existing boilers can be also cost-effective. For this replacement new retort and counter current boilers are recommended. These measures should be coupled with general improvement of building energy performance. The replacement can be done at the natural end of the boiler's service life, thereby minimising any additional costs.

The implementation of technical measures requires public support through financial incentives for home owners (e.g. tax deductions, low rate loans, etc). There are examples in BSR of these kind of instruments successfully implemented. Residential sources are most relevant on a regional level as air deposition is limited to the emission sources (>PM10). Moreover, a mix of measures should be tailored to regional problems including awareness raising, financial support schemes, strengthening regulations and enforcement. The process of improvement will need at least 20 years time for gradual household investments .

For industrial air abatement, technical measures with improved combustion and clean up techniques are the most cost-effective. The action on an European level will be the most efficient as the reduction both in BSR countries and the rest of Europe can be achieved. Improvement of BREF for energy production, metallurgical sector, and waste incineration would provide emission reductions of up to 50%.

For open burning and residential combustion, municipal waste handling and disposal – strengthening of the waste collection system through appropriate country and local regulations combined with awareness raising would prevent improper conduct of household owners and SMSs.

Contaminated land and sediments is a regional problem and should be treated on a case by case basis. For the high risk sites, high contamination levels should be treated with technical measures especially dredging and on site combustion of contaminated wastes or soil. On the basis of Finnish examples on–site capping is also recommended.

Because dioxins and furans persist in the body for years, recent significant reductions of emission into the air are unlikely to reduce overall human health and environmental risks to the general population in the near term. From a whole BSR perspective no universal approach can be recommended as country specific problems differ widely.

2. Organotin compounds: Tributyltin compounds (TBT) and Triphenyltin compounds (TPhT)

More research on the state of Baltic Sea contamination with organotins might be helpful for deciding the best techniques of cleaning the sea, as for the moment the available data on this issue is rather old or not of relevance.

Although a great number of different EU regulations and directives defining the use of organotins exist, national legislation considering technical possibilities of implementation and linkage between these regulations is still needed. Historical pollution and pollution from articles that is produced outside the EU must be dealt with.

All methods for eliminating TBT/TPhT from the Baltic Sea seem to be costly but at the same time highly effective. The reduction of organotin quantities in water due to different bans on the use of TBT/TPhT should be taken in mind when deciding which method for elimination of TBT/TPhT should be chosen.

3. Polybrominated Diphenyl Ethers (PBDE)

PBDEs are a group of substances which are used commercially as flame retardants in specific product groups, especially in plastic parts of electrical appliances. The legal system on hazardous substances management currently in force prohibits the manufacture and use of most of commonly used PBDE derivatives or strictly regulates their use. As a result, production of this substance group is decreasing steadily worldwide. However there still are some uses of BDEs persisting especially as for decaBDE. As most of the emissions are generated during the production and service life of different consumer products, the most effective measures to reduce total emissions address the substance at source – either by substituting it with alternative flame retardants (with a large variety of affordable substitutes existing) or by managing waste water flows from production facilities.

A product group mentioned in substitution chapter but not covered in measures assessment is construction materials where PBDEs are also used, however, this sector for PBDEs is rather insignificant and being “dominated” by other brominated flame retardant – hexabromocyclododecane. As a result, measures in this production segment would result in rather small reduction of PBDE emissions.

Historical pollution from the use and dismantling of previously produced articles also must be dealt with – as this is mostly diffuse pollution, adequate steps must be taken to manage the matrix of final accumulation – sludge from waste water treatment plants. For heavily contaminated water bodies, where serious problems with historic contamination persist, sediment management – environmental dredging -can be considered as an option.

4. Perfluorinated compounds (PFCs): perfluorooctanesulfonate (PFOS) and perfluorooctanoic acid (PFOA)

From a single-substance specific perspective, the most cost effective measure for reducing PFOS emissions was determined to be substitution by polyfluorinated drop in systems in metal (chromium) plating.

As these drop in substitutes are available and are markedly less toxic and bioaccumulative than PFOS, there is no reason that PFOS should continue to be exempted from the EU ban for this application. But even though they are the most cost effective measure, polyfluorinated substitutes can only be a short term solution, as they also give cause for environmental and health concerns. A more sustainable long term solution would be to additionally introduce appropriate end of pipe measures, such as AC filters, to reduce emission of polyfluorinated substitutes. These end-of-pipe measures for emission reduction can be implemented by improving BAT and revising the metal surface treatment BREF, which is one of the recommendations of this guidance document.

For PFOA, substitution of PFOA in manufacture of semi conductors was found to be the most cost effective measure, especially if a single (combined) substitute can be developed for PFOS/PFOA. As this is an emerging measure, the uncertainties are high and development time is needed before the measure can become effective. As these future substitutes will likely be based on polyfluorinated substances, the same reasoning as above applies. In case of high emission factors, appropriate end of pipe measures should additionally be researched and applied to avoid high loads of the substitutes to environment.

Besides measures at industrial sources, measures at urban sources can also reduce emission of PFOS/PFOA, such as advanced treatment of municipal waste water by activated carbon. Even though this technology is available and proven, its cost effectiveness is subject to high uncertainties. The cost effectiveness of advanced treatment of municipal waste water is strongly dependent on the combined load of hazardous substances per person and year in incoming municipal waste water and the removal efficiency of the measure.

Therefore, before introducing advanced waste water treatment, the combined load of hazardous substances per person should be assessed on individual facility level for large MWWTPs to decide whether the cost effectiveness of the measure is competitive. It can be expected that the combined load will vary for different urban areas, due to different user behaviour and different patterns of indirect dischargers. But on the other hand, if high loads of one hazardous substance are found in incoming municipal waste water, analysis of indirect dischargers to this MWWTP is advisable. Generally, it is more cost effective to reduce emissions close to their industrial sources. In the presented example, the factor for PFOS is in the range of 1000-10 000.

From a single substance specific perspective treatment at MWWTPs tends to be more costly per kg of reduced emissions than treatment at the industrial sources themselves, especially for PFOS. However, a cross substance perspective is needed for evaluation, as there are many hazardous substances in municipal waste water and activated carbon has a broadband effect. Therefore, when taking into account the effect on all 11 hazardous substances, it may be worthwhile to consider making advanced treatment of municipal waste water an element of a cost effective strategy for reducing emissions of hazardous substances in the BSR. This issue is discussed in COHIBA recommendation report.

The described measures for reduction of PFOA and PFOS emissions should be flanked by awareness raising measures for the public and enterprises, since these measures are relevant for all 11 hazardous substances which are the focus of BSAP.

Cost effectiveness is an important criteria for selection of measures for emission reduction, but other criteria, such as secondary environmental effects, technical feasibility or political enforceability are also important. The data has inherently large uncertainties. As accurate information is not available, the evaluation is based on approximations of size of sources, emission factors, effectiveness and costs of measures.

One of the main challenges in evaluating reduction measures is the lack of reliable and up to date information on loads to the environment. As accurate information is not available, the evaluation is based on approximations of size of sources, emission factors, effectiveness and costs of measures. Obtaining accurate information is hindered by dynamic change in emission patterns due to recent regulation of PFOS and issues with confidential business information (CBI). For PFOA, there is even less information available especially on industrial sources and corresponding emission factors. PFOA is not regulated at EU or international level (see Chapter 4), therefore there are also no reporting duties. This regulatory gap for PFOA should be addressed.

5. Hexabromocyclododecane (HBCDD)

There are a number of measures available to reduce emissions of HBCDD in the Baltic Sea Region. Cost-effectiveness was calculated for the prioritised measures (where data were available). Uncertainties exist in all results, which means that the data presented in this guidance document should be interpreted with caution.

Of the alternatives studied here, use of non-flame retarded EPS and XPS or mineral wool seems to be the most cost-effective option for substituting insulating boards flame retarded with HBCDD in the building sector. Mineral wool cannot however be used in all applications in which polystyrene boards are used. For applications where structural stability is needed, PUR foam could be an alternative option.

Mechanical, chemical and biological waste water treatment appears to be an efficient measure to remove HBCDD from wastewater. The Urban Waste Water Directive provides that waste water must be collected and treated in all agglomerations of >2000 population equivalents (p.e.). Besides having a high efficiency, waste water treatment also has several positive secondary effects such as reduction of nutrient loads to receiving waters as well as removal of a wide range of hazardous substances from effluent. In addition to mechanical, chemical and biological waste water treatment, it is also possible to further treat the waste water using advanced treatment options. One example of such a measure is activated carbon treatment. By additionally implementing these measures, the removal of HBCDD can be further increased.

Among the end-of-pipe measures, anaerobic digestion of sewage sludge is seemingly the most cost-effective alternative to be implemented at a municipal WWTP. However, this measure reduces HBCDD levels to a lesser extent compared to the measure that also uses thermal hydrolysis.

Site specific separation of sludge at industries is cost-effective in reducing HBCDD levels in effluent waters. The substance is however only partitioned to the sludge and the industry will have additional costs for sludge treatment.

It would also be of interest to further study the cost-effectiveness of waste management measures, such as demolition of building as this is predicted to be one of the major sources of HBCDD emissions in the future.

6. Nonylphenol (NP) and Nonylphenol ethoxylates (NPE)

In general, industrial sources seem to be much more significant than urban sources of total NP(E) emissions to the environment but the situation is to some extent different in each Baltic Sea country. Urban emissions are channelled through urban infrastructure systems (MWWTPs, waste disposal/incineration sites as point sources), or through diffuse urban emissions (urban run-off). Municipal waste water treatment plants (MWWTPs) are the most significant urban source of NP(E). Service life plays only minor role as emission source to environment. Nevertheless, the service life (especially the use of cosmetic and hygiene products & washing of textiles) is an important upstream source to MWWTPs.

Several NP and NPE applications are only conditionally banned (use allowed if no emissions) in EU for industrial cleaning, metal working, textile and leather processing.

Thus, it can be concluded that EU wide absolute ban of NP and NPE is primarily recommended for the following usages:

- Industrial and professional (or institutional) cleaning (manufacture, marketing & use); conditional ban to be replaced by absolute ban
- Textiles (manufacture, marketing, use & import of articles); conditional ban to be replaced by absolute ban & import of textile articles containing NP/NPEs to be banned

Secondarily, the EU wide absolute ban of NP and NPEs is recommended for the following usages:

- Leather (manufacture, marketing & use)
- Metal working (manufacture, marketing & use)

These measures would significantly facilitate authority control and cessation of emissions from the abovementioned significant NP and NPE emission sources. Currently national chemical product registers of certain Baltic Sea countries (registers available at least in Denmark, Finland and Sweden) indicate that NP and NPE are still used in certain conditionally banned usages (e.g. cleaning) but it is not possible to draw unambiguous conclusions whether emissions to waste water or environment occur or not. After an absolute ban of NP and NPE in abovementioned usages, authorities can draw conclusions that regulations are not obeyed by companies using NP and NPE in certain usages (e.g. cleaning sector) based on Chemical Product Register data. Then authorities can immediately start control measures against these companies.

The alcohol ethoxylates are cost-effective and already used substitutes for NP and NPE in above mentioned applications.

A ban on NP and NPE in paints would not be as cost-effective as banning their use in cleaning, but still moderately cost effective, especially since the amounts used in some Baltic Sea countries are high.

For waste waters (including leachate) with concentrations of NP(E) above the EQS (0.3 µg/l) value, the use of End-of-Pipe techniques like AC is recommended. Since AC treatment reduces several pollutants simultaneously, it seems to be a cost-effective measure compared to other End-of-Pipe techniques. Additionally, AC treatment is a proven technology with quite a long lifespan for usage over 80 years.

Since sewage sludge is the most important receiving compartment of NP(E) in waste water treatment plants, a proper disposal of sludge is essential. NP(E) is quite easily degradable under anaerobic and aerobic conditions which makes it quite easy to treat the sludge firstly through anaerobic degradation and secondly through aerobic composting.

Urban run-off can also be a potential NP(E) source to the Baltic Sea coastal areas and a proper handling is recommended if the EQS value is exceeded. Concerning urban run-off treatment, sand filtration systems seem to be a cost-effective measure. The large handling capacity and the wide cross substance effects are further advantages of this method.

To sum up, concerning only NP and NPE source control and regulatory measures like banning and the subsequent substitution with alcohol ethoxylates are posing the most cost-effective reduction potential and are highly recommended. A ban should be primarily targeted in the cleaning and textile sectors and secondarily in the leather and metal working sectors. Thus, those usages which were only conditionally banned in 2005 (directive 2003/53/EC) are recommended to be totally banned. Additionally, the ban of NP and NPE in textiles imported from outside of EU is recommended. End-of-Pipe measures are recommended in cases where EQS values of NP in effluents, landfill leachate or urban run-off are exceeded or when there is also the need to reduce other harmful substances from waste waters at the same time.

7. Octylphenols (OP) and octylphenol ethoxylates (OPE)

Octylphenols and octylphenol ethoxylates are widely used in very different applications. The main use of OP is resins in tyre manufacture, so for safety reasons it is not possible to ban OP completely.

Nevertheless, the chemical industry in Europe is already prepared for a more restricted use of OP and OPE, as the voluntary agreement of industry in the UK shows. The producers of chemical specialities invested in research and development (R&D) of environmentally friendly, but functionally equivalent, alternatives to OPE. Substitutes like e. g. fatty alcohol ethoxylates have additional costs of only 10- 20 %.

The downstream industry, which produces consumer products, has to be pushed to use the substitutes. This can preferably be done with voluntary agreements, which can be adjusted more easily to the market and technical framework conditions of each industrial sector. It might be a challenge to implement voluntary agreements on BSC level.

For OP and OPE which are already in use in products, two end-of-pipe measures for MWWTPs were analysed: “Advanced waste water treatment – AC treatment” and “Sludge treatment - controlled incineration”. Both measures have cross substance effects and are capable of eliminating other hazardous substances.

8. Short-chain chlorinated paraffins (SCCP) and medium-chain chlorinated paraffins (MCCP)

Use of SCCP in the two most relevant former uses has been restricted at EU level (metal working and fat liquoring in the leather industry). However, SCCP containing products (e.g. SCCP containing leather) are still in use and will continue to end up as waste over the coming years. Due to additional voluntary approaches (PARCOM 95/1) use of SCCP has also further decreased in all other industries in the EU and substitutes have been found for many areas of application. Nevertheless, SCCP are still used in the rubber industry (about 80% for conveyor belts), sealants, paints & varnishes and textiles (all products have lifetimes of several years).

While SCCP containing products end up as waste after their service life, diffuse emissions from products containing SCCP during their service life mainly enter waste waters or other compartments e.g. via atmospheric deposition.

The main measures for the waste water compartment are end-of-pipe measures at waste water treatment plants. While conventional wastewater treatment does not seem to be effective enough to achieve significant reductions of SCCP emissions to the environment, it can be assumed that advanced waste water treatment such as activated carbon filtration, membrane filtration or advanced oxidation processes will have a significant effect.

SCCP are assumed to mostly adsorb onto sludge during waste water treatment resulting in relevant concentrations in sewage sludge (however, probably 95% of sludge is only contaminated little and 5% is higher contaminated). Therefore controlled sludge treatment instead of application of sludge to agricultural land can be considered as an additional measure.

For the remaining products containing SCCP, appropriate waste management (e.g. controlled collection and incineration) could be applied as measure for reduction of SCCP emissions. For these reduction measures the relevant main characteristics of SCCPs are their high adsorption capacity (adsorption onto sludge -> sludge treatment), and a decomposition at temperatures above 200°C. Consequently, controlled incineration of sludge and waste is an effective measure. However, due to valuable nutrients contained in sludge (phosphorus) sludge and waste should be incinerated separately (mono incineration).

As additional measures restrictions on the use of SCCP could be extended to include the remaining areas of application (rubber, paints & varnishes). More important, since substitution of SCCP has proven to be possible in most areas of application, the knowledge regarding substitution should be transferred to regions outside EU where SCCP are still produced and used in large amounts (reduction of immissions via long range transport).

The key messages regarding measures to reduce MCCP emissions are similar to those for SCCP. Possible restrictions and substitution of MCCP need to be further evaluated, since industry and the use of products containing MCCP seem to be the dominant sources of MCCP emissions to the environment. The use of MCCP as substitute for SCCP should be avoided.

Again appropriate waste management (collection, controlled landfilling or incineration) could be applied to prevent distribution of MCCP via the pathways ‘products’ and ‘waste to the environment’.

The main measures for reducing MCCP emissions via waste water are end-of-pipe measures at waste water treatment plants. While conventional waste water treatment might not be effective enough to achieve significant reductions of MCCP emission to the environment, it can be assumed that advanced waste water treatment practices such as activated carbon filtration, membrane filtration or advanced oxidative techniques will have a more significant effect.

Controlled incineration of sludge could be applied as additional measure instead of agricultural use of sewage sludge to avoid distribution of MCCP via MCCP containing sludge. However, data on MCCP concentrations in sludge are scarce; therefore additional measuring campaigns need to be conducted.

9. Endosulfan

Endosulfan has been identified as a priority hazardous substance under the EQS Directive and banned for manufacturing and use in the EU. Because of its threats to the environment, a global ban on the use and manufacture of endosulfan is being considered under the Stockholm Convention[1]. Endosulfan is likely, as a result of its long-range environmental transport, to lead to significantly adverse human health and environmental effects so that global action is warranted. Therefore the main measure for endosulfan is a global ban.

Part of endosulfan emissions are emitted through diffuse sources: imported foodstuffs from countries still using endosulfan as a plant pesticide, emissions during the manufacturing of food products and from household consumption of foodstuffs. Due to the physical and chemical properties of endosulfan, the substance binds to the solid fraction in the aquatic environment. Thus it is crucial to correctly manage the matrix where most of the substances accumulate – sludge of WWTPs, where controlled landfilling or controlled incineration of sludge containing endosulfan is the most effective measure. Concerning advanced wastewater treatment, activated carbon filtration is an effective measure showing an effectiveness of removal of 75-98%.

Although the use of endosulfan is banned in the EU, historical pollutions occur e.g. in wastewater sludge and landfill sludge in countries which used endosulfan in earlier years. Thus, cleaning up contaminated sites and management of obsolete pesticides may become a particularly relevant issue in countries where endosulfan is manufactured.

Improving the control of and banning contaminated foodstuff is an important precautionary measure because contaminated foodstuff does not only originate from developing countries but also occurs in countries within the EU – despite the existing ban. Therefore, further investigations in terms of different sources and their respective importance are needed.

10. Mercury (Hg)

Mercury is a naturally occurring element and is subject to a number of European regulations. It is still used in various industries and in common consumer products. At present, some mercury applications are restricted or even banned.

The main route of water contamination with mercury is the direct and indirect discharge from point sources, including the use of dental amalgam, chlor-alkali plants, and power plants, ferrous and non-ferrous industries along with waste disposal. There are several technological measures to avoid mercury emissions described as the Best Available Technologies (BAT). The most effective measures to reduce mercury emissions to air are fabric filters and adsorption using activated carbon/activated lignite coke.

The main options to reduce mercury emissions to the environment include source control and end-of-pipe solutions in water treatment technologies. These sources have direct impacts on the aquatic compartment from point sources, and indirect impacts via atmospheric deposition and leaching from landfills.

Substitution is very important for the use of products containing mercury, e.g. batteries, electrical and electronic equipment and thermometers, due to their life cycle impacts. Substituting mercury in dentistry is the most highly recommended reduction measure.

11. Cadmium (Cd)

Upgrading and retrofitting of burners in households is potentially the most efficient measure on the catchment area scale, especially in Poland. Unfortunately the implementation of this measure would require 20 years since gradual investments by house owners are needed with support actions on the part of the public sector. The measure is linked directly to energy efficiency improvements in the residential sector.

Improvements in dust emission in the power sector, waste incineration and metallurgy are cost-effective for certain techniques applied in industry where the baseline technical

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[1] At its fifth meeting held from 25 to 29 May 2011, the Conference of the Parties adopted an amendment to Annex A to the Stockholm Convention to list technical endosulfan and its related isomers with a specific exemption (decision SC-5/3). Pursuant to paragraph 4 of Article 21 of the Convention, the amendment was communicated by the depositary to all Parties on 27 October 2011

environmental standard is relatively low. The cost-effectiveness is moderate when additional investments are required.

The abatement techniques available today can lead to a further 50% reduction of emissions in industry. This measure is in line with new proposed EU regulations cutting the dust emissions by 50%. 10 years time can be assumed as the time of implementation. The particulate abatement techniques are also have a positive effect on the reduction of other emissions e.g. dioxins and furans and PAH.

An additional reduction of cadmium emissions in waste water has a comparatively low potential on the catchment area scale but it is still important in selected industrial sectors/facilities and municipal WWTPs. These problems should be tackled more on a case by case basis.

Measures tackling cadmium emission from products are assumed to have low reduction potential as the cadmium use is banned in most of the general applications. Nevertheless, raising public awareness is a good option for local communities or certain sectors, e.g. energy or low level emission sources with consumer use – for example semi-professional artists, car workshops, dismantling facilities for electronic equipment.

Improvements in waste management systems are important as many products will become waste in the near future and recovering cadmium from the stream will prevent additional input as diffuse pollution. The measure will be important for the next 10 –20 years when the products containing cadmium will go into the waste. Bans on use of cadmium in products will lower the emission from general consumption and waste processing. For 20 years the recycling of waste and control will be needed to contain the cadmium in a safe manner. Good enforcement of regulatory requirements and voluntary schemes of material recovery are cost-effective measures in preventing an improper disposal of waste containing cadmium.

The treatment of contaminated land is important for selected regions were the soil and sediment pollution is high.

B. Comparison of reduction measures

Key

+	Only limited effectiveness	Very high costs	Negative secondary environmental effects	Technology not yet available or very new management option	Negative or no socio-economic effect	Only long- term realization, > >10 years	Strong political opposition expected	Costs per kg (or per Teq) emission reduction high
++	Partially effective	Moderate costs	Several positive secondary environmental effects	Pilot process or transferrable non-technical measures available	Some positive socio-economic effects	Medium-term realization, approx. 3 - 10 years	Political opposition expected	Costs per kg (or per Teq) emission reduction medium to high
+++	Substantial effects	Low costs	Numerous positive secondary environmental effects	proven and available technology	Many positive socio-economic effects	Rapid realization possible, 1-3 years	Political support expected	Costs per kg (or per Teq) emission reduction medium to low

1. PCCD/F	Effectiveness	Cost	Secondary environmental effects	Technical feasibility	Secondary socio-economic effects	Geographical/time scale of effects	Political enforceability	Cost effectiveness
Measure 1: Replacement and retrofitting of household heating furnaces	+++	++/+++ private	+++	++/+++	++	++ PL, DK, RU	+	++ private
Measure 2: Financial support programs for residential sector emission reduction	+++	++/+++ private	++	+++	+++	++ PL. RU	+	+++ public
Measure 3: Public awareness raising	++	+/+++	++	+++	++	+++ EBSR	+++	+
Measure 4: Waste management	++	+++	++	++	-/+	+ PL. RU	+/++	+++
Measure 5: Improvement of BAT and revision of BREF document concerning dust emission reduction	++	-/+	+++	+++	+	++	+++	++ (1-50 MW)
Measure 6: Treatment of contaminated soil	++	+/+++	++	+/++	+/++	? ALL	+	BS+/+++regional scale
2. TBT/TPhT	Effectiveness	Cost	Secon-dary environmental effects	Tech-nical feasibility	Secon-dary socio-economic effects	Geographical/time scale of effects	Political enforceability	Cost-effectiveness
Measure 1: Sediment management - environmental dredging	++	+++	++	+++	-/+	++	+	+
Measure 2: Sediment management – no dumping of substance TBT/TPhT sediments in the Baltic Sea	+++	+++	+	+++	++	++	+	-
Measure 3: Sediment management – ex situ	++	++	+	+	-/+	+	+	+

solidification and stabilisation								
Measure 4: Sludge treatment – controlled incineration	+++	+++	+	-/+	+	+	+	-
Measure 5: Waste management – controlled land filling	+++	+++	-/+	-/+	-/+	+	+	-
Measure 6: Advanced waste water treatment – AC treatment	+++	+	+	+	++	++	++	+++
3. PBDE	Effectiveness	Cost	Secondary environmental effects	Technical feasibility	Secondary socio-economic effects	Geographical/time scale of effects	Political enforceability	Cost-effectiveness
Measure 1: Substitution of PBDEs in polymers, textiles and construction materials:	+++	++/+++	++	+++	++	++/+++	++	++
Measure 2: Sludge management – controlled incineration:	++	+	+++	++	+	++	+	+
Measure 3: Urban run-off management:	++	+	+++	++	+	++	++	+
4. PFOS and PFOA	Effectiveness	Cost	Secondary environmental effects	Technical feasibility	Secondary socio-economic effects	Geographical/time scale of effects	Political enforceability	Cost effectiveness
Measure 1a: Substitution of PFOS in metal (chromium) plating	+++	+++	-/+	+++	+	+++, DE, DK, . SE, FI	+++	+++
Measure 1b: Substitution of PFOS/PFOA in semi-conductor industry	+++	+	-/+	+	+	++	+++	+
Measure 1c: Substitution of PFOS/PFOA in photographic industry	+++	+	-/+	+	+	++	+++	+
Measure 2: Improvement of BAT and revision of BREF document for metal surface treatment	+++	+++	+	+++	+	+++ DE, DK, . SE, FI	+++	++
Measure 3a: Advanced waste water treatment - AC treatment for industrial waste water	+++	++	++	+++	+	+++ DE, DK, . SE, FI	+++	++
Measure 4: Advanced waste water treatment - AC treatment for municipal waste water	++	+	+++	+++	+	++	+	+
Measure 5: Public awareness raising	?	++	+++	+++	+	++	+++	?
Measure 6: Awareness raising for enterprises	?	++	+++	+++	+	++	+++	?

5. HBCDD	Effectiveness	Cost	Secondary environmental effects	Technical feasibility	Secondary socio-economic effects	Geographical/time scale of effects	Political enforceability	Cost effectiveness
Measure 1: Substitution of HBCDD as a flame retardant	+++	+++	No data	+/+++[2]	No data	No data	No data	+++
Measure 2: Changing of product material	+++	+++	No data	+++	No data	No data	No data	+++
Measure 3/4: Mechanical, chemical and biological waste water treatment (industrial/municipal waste water)	+++	No data	+++	+++	No data	No data	No data	No data
Measure 5: Sludge treatment – anaerobic digestion	++	+	++	+++	No data	No data	No data	+
Measure 6: Sludge treatment – thermal hydrolysis and anaerobic digestion	+++	+	++	++	No data	No data	No data	+
Measure 7: Waste management – controlled incineration	+++	No data	+	+++	No data	No data	No data	No data
Measure 8: Waste management – controlled landfilling	++	No data	+	+++	No data	No data	No data	No data
Measure 9: Waste management – demolition of buildings	No data	No data	No data	No data	No data	No data	No data	No data

6. NP and NPE	Effectiveness	Cost	Secondary environmental effects	Technical feasibility	Secondary socio-economic effects	Geographical/time scale of effects	Political enforceability	Cost effectiveness
Measure 1a: Substitution of NPE in cleaning	+++	+++	None	+++	+	++ All Baltic states	+++ (EU member states)	+++
Measure 1b: Substitution of NP and NPE in paints	+++	++	None	+++	+	++ All Baltic states	+++ (EU member states)	++
Measure 2: Restrictions on marketing, use and import of textiles	+++ 2	+++ 2	None	+++	None	++ All Baltic states	+++ (EU member states)	+++ 2
Measure 3a: Advanced waste water treatment – active carbon treatment (AC)	+++	+(+)	+++ (removes also	+++	+	+++[3]	++ (EU member	++

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[2] Substitution of HBCDD in EPS and XPS will soon be possible, but substitutes already exist for its use in HIPS plastic and for textile back-coatings.

			other hazardous substances and nutrients)				states)	
Measure 3b: Advanced waste water treatment – membrane filtration (NF/RO)	+++	+	+++ (removes also other hazardous substances and nutrients)	+++	+	+++3	++ (EU member states)	++
Measure 3c: Advanced waste water treatment – oxidation	++	+	+++ (reduces in general organic matter hazardous substances load)	++	+	+++3	++ (EU member states)	+
Measure 4: Urban run-off treatment	+++	++	+++ (reduces in general nutrient & hazardous substances load)	++(+)	+	++3	++ (EU member states)	++(+)
7. OP/OPE	Effectiveness	Cost	Secondary environmental effects	Technical feasibility	Secondary socio-economic effects	Geographical/time scale of effects	Political enforceability	Cost effectiveness
Measure 1: Substitution of OP in textile printing	++	++	+	+++	+	+++	+++	+++
Measure 2: Incineration of waste tyres	++	+	++	+++	+	++	++	+++
Measure 3: Voluntary agreement to stop using OP	+++	++	++	++	++	++ (not RU)	+++	+++
Measure 4: Advanced waste water treatment – AC treatment	++	++	+++	+++	+	+	++	+
Measure 5: Advanced sludge treatment: Controlled incineration	++	++	+++	+++	+	++	++	++

8. SCCP/MCCP	Effectiveness	Cost	Secondary environmental effects	Technical feasibility	Secondary socio-economic effects	Geographical/time scale of effects	Political enforceability	Cost effectiveness
Measure 1: Ban on SCCP in remaining areas of application Banning/restricting the use of MCCP in emulsion and oil based metal cutting fluids	++	++	-	++		+	++	
Measure 2: Substitution of SCCP in rubber products and substitution of MCCP in emulsion and oil based metal cutting fluids	+++	++		++		+	++	++
Measure 3, 4, 5: Advanced waste water treatment (activated carbon (AC) treatment, membrane filtration, oxidative techniques)	++	++	+++	++		++	++	++
Measure 6: Sludge treatment – controlled incineration	++	++	+++	+++		++	++	++
Measure 7: Waste management – controlled landfilling / controlled incineration	++	++	+++	+++		++	++	++

9. Endosulfan	Effectiveness	Cost	Secondary environmental effects	Technical feasibility	Secondary socio-economic effects	Geographical/time scale of effects	Political enforceability	Cost-effectiveness
Measure 1. Ban and substitution of endosulfan	+++	+	+++	++	++	+++	++	++
Measure 2. Waste management – controlled land filling / controlled incineration	+++	++	+++	++	+++	++	+++	+++
Measure 3. Advanced waste water treatment – AC treatment	++	+	+++	+	+	++	++	++
Measure 4. Improvement of control and ban of contaminated foodstuff	+++	++	+++	++	++	++	+++	+++

10. Mercury	Effectiveness	Cost	Secondary environmental effects	Technical feasibility	Secondary socio-economic effects	Geographical/time scale of effects	Political enforceability	Cost effectiveness
Measure 1: Substitution of mercury in products and processes								
Dentistry materials substitution	+++	++	+	++	++	++	+++	++
Electric products substitution	+++	++	+	++	++	+++	+++	++
Measure 2: Improvement of BAT and revision of BREF								

for combustion power plants								
Fabric filters	+++	++	+++	++	++	+++	++	++
Electrostatic precipitators	++	+++	++	+++	++	+++	++	++
Adsorption using activated car bon/activated lignite coke	+++	+	++	++	++	++	++	++
Measure 3: Waste management - controlled incineration								
Waste incineration	++	+++	+++	++	++	+	+	++
Crematoria emission reduction techniques	+++	+	+	+	+	++	++	++

11. Cadmium	Effectiveness	Cost	Secondary environmental effects	Technical feasibility	Secondary socio-economic effects	Geographical/time scale of effects	Political enforceability	Cost effectiveness
Measure 1: Improvement of BAT and revision of BREF document concerning industrial air abatement	++	+/++	++/+++	+++	++	+/++ *5-10 years all*	+++	+++
Measure 2: Improvement of BAT and revision of BREF document concerning wastewater treatment	+/++	++	++	+++	++	+++*PL, BSR 5 years*	+++	++ *regional*
Measure 3: Replacement and Retro-Fit of household heating furnaces	+++	+/++	+++	+++	++ *financial*	+/++ *10-30 PL, DK, RU*	+	+/++ *regional*
Measure 4 Waste management - recycling	+/++	+++	+++	+++	++	++ *5-10 years RU. PL*	++/*partially exists*	+/++ *regional*
Measure 5: Reduction of Cd content in fertilisers	+++	+*fertilisers*/+++*sludge*	++	+ *fertilisers*/+++*sludge*	++ *depend on the culture*	+	*PL, EBSR, RU*	+/+++ *organic matter use, regional*
Measure 6: Advanced waste water treatment - AC treatment	+++	++	+++	+++	+/++	+ *PL RU*	+++	+
Measure 7 Public awareness raising	++	+++	+	+++	++	+*all*	+++	+/++ *sectoral*
Measure 8: Treatment of contaminated soil	+++*local*	++	+	++	+	+ *RU, LT, LV, EE*	+/+++	++ *local*

C. Detailed substance-specific assessment of measures

I. DIOXINS, FURANS AND DIOXIN LIKE POLYCHLORINATED BIPHENOLS

1.1 Measure 1: Replacement and retrofitting of the household heating furnaces

1.1.1 Description of source

This measure concerns individual households with boilers 10-100 kW both in detached houses and apartment buildings, medium size energy plants 100 –1 MW operated by Small and Medium Enterprises (SMEs) and central heating plants < 1 MW owned by private entities, public or housing establishments. Low efficiency of burners, low fuel quality and waste burning in installations causes high emissions of PCCD/F[3]. This source constitutes 20% of air deposition to Baltic Sea catchment with further impact on soil contamination. This source has the biggest reduction potential in Poland and Russia.

1.1.2 Description of measure

Emission is reduced by retrofitting and modernisation of household heating furnaces, housing infrastructures like central heating systems, flue gas treatment utilities, and fuel change with sound renewable energy use. There are a variety of technical solutions related to solid fuels, burning installations, conditions in the burner chamber and flue gas scrubbing.

1.1.3 Effectiveness

Technical efficiency of emission reduction on the household level is high. Up to 95% reduction in comparison with low efficient utilities commonly used for example in Poland.

1.1.4 Costs

Costs vary according to technical solutions and conditions[4]. The turnover time for domestic heating systems is slow and technical solutions are few and relatively costly. Investment costs for new advanced installation are 1000-2000 Euros and 400-500 Euros for simple furnaces with high dust emission. Additional costs for auxiliary installations, montage and service should be included (20% on top). Operational costs for house owners depend on maintenance and fuel costs. For example coal culm as the lowest grade coal is used widely in Poland due to social and economical (4 -5 Euro/GJ) reasons. The high quality fuel including high grade coal is 20- 50% more expensive (5,5-6,5 Euro/GJ). It should be noted that the advanced installations can reduce the fuel consumption by up to 40%, which can be further reduced by thermo-insulation by additional 30%. Effectiveness of emission reduction is around 3-8 Euro per ug I-TEQ (140 ug I-TEQ per household). Costs of air abatement measures are comparable, e.g. investment costs for Electrostatic Precipitators (ESP) is 20 Euro per kWth with dust emission reduction cost from 420 Euro (80% efficiency) to 15000 (95%) Euro per Mg of dust (mg I-TEQ). For advanced boilers ESP is not economical.

1.1.5 Secondary environmental effects

In the case of dust removal, waste (fly ash) is produced and for automatic boilers additional electricity is used. The energy demand in production of different fuels should be taken into account e.g. in processing of biomass and coal into briquettes and pellets, the energy loss is about 20%. Positive aspects are a reduction of fine particular dust emissions, Green House Gases (GHG) emissions and improved health conditions.

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[3] Installations not covered by IPPC and IED Directives

[4] It should be noted that for complex modernisation schemes (buildings, on-site biomass use) the investments are economically attractive with a few years payback on energy savings. Costs of thermoinsulation in Poland for building of 100 m^2 space is around 8000 Euro with the yearly savings around 50% of heating costs and reducing the PCCD/F emissions.

1.1.6 Technical feasibility

Reduction of residential combustion emissions can be achieved through replacement or improvement of individual combustion installations:

- Improvement in combustion processes (chimney insulation, seeping, thermo catalysis, accumulator tanks). Good maintenance practice is basically well known.
- Efficient small burners for high quality fuels include: retort; counter current; and automatic boilers (BIPRO, 2009). These advanced burners are commercially available and are offered by many competing producers.

Technical measures for air emission reduction, including dust filters, are feasible but are rarely used in medium and small installations. There is also potential in all BSR countries for producing high quality fuels.

1.1.7 Secondary socio-economic effects (incl. indirect costs)

These are incurred by house owners or public entities. The cost of heating is an important share in the household budget in Poland and other EBSR countries. A positive aspect is the reduction of health risk.

1.1.8 Geographical and time scale of effects

The issue is most relevant for PL 60% of residential emissions, RU 28 %, DK 4%, less important in EE, LV, SW, DE, 1 – 2 %. The time frame of these measures taking full effect varies between 10 - 30 years depending on investments and establishment of good practices. Public support policies are also an important time factor of implementation.

1.1.9 Political enforceability

The measure can be implemented on voluntary basis. In general it is politically feasible but it depends on the country and local specific situation (low and medium feasibility). There is a set of instruments applied in the BSR countries: technical standards, regular inspections. The potential benefits are not recognised sufficiently. The measure is complementary or competing to other measures e.g. district central heating. Thermal insulation of buildings is also an important factor for final efficiency and technique selection5. The measure requires standardisation or certification schemes for boilers and solid fuels including biomass pellets and briquettes and coal briquettes e.g. biomass standards exist but should be widely recognised6.

1.1.10 Cost-effectiveness analysis

Residential combustion has only 54% respectively 20% PCDD/F reduction potential in relation to total PCDD/F air deposition to the Baltic Sea catchment area and 34% to the Sea but it also causes long term secondary contamination (COHIBA estimations). Reduction potential is regional and problem specific e.g. in Poland the new automatic retort boilers amounted to 4% of the total solid fuel use in residential sector. According to an estimation made for Polish cities, 25% of PCDD/F emission reduction between 2006 and 2020 is a realistic target. For other countries the reduction potential is lower. The total reduction (sector scale) for the BSR can be estimated at 60% to 90% (COHIBA data) comparing it with the current situation in the relevant countries e.g. in Poland and it depends on awareness of house owners, financial capabilities or technical barriers in buildings. Reduction of PCDD/F per year for the cost-efficient scenario is 150 grams of I-TEQ per year (Polish case). The value can be roughly doubled for the whole region.

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[5] In principle the thermoinsulation is a prerequisite of the heating system modernisation. For complex modernisation schemes (buildings, on site biomass use) the investments are economically attractive with a few years payback on energy savings. Costs of thermoinsulation in Poland for building of 100 m^2 space is around 8000 Euro with the yearly savings around 50% of heating costs.

[6] Reduction of dust in fuel below 5% and chlorine below 0.1% cause the reduction of PCDD/Fs by over 80% in boilers with automatic burning process and about 20% in boilers with manual fuel feeding.

Cost-effectiveness for the catchment scale varies between low to medium. Total costs calculated for Poland are between medium (improvements) to high 5 – 10 billion Euro (investments). For Russia, and other countries the estimation can be doubled. Yearly cost of fuels in Poland can be estimated at 2 bn Euro/year in the most cost–efficient case. In Finland, installation of small Electrostatic Precipitators (ESPs) with marginal costs between 420 –3700 for simple boilers (logs) can give reduction of 2 Gg PM2.5 (g PCDD/F). Installation of accumulator tanks can give 1,5 Gg PM2.5 reduction. These are ten times lower than for industrial facilities (Karvosenoja, 2007).

1.2 Measure 2: Financial support programs for residential sector emission reduction

1.2.1 Description of source

This measure focuses on individual households with boilers 10-100 kW both in detached houses and apartment buildings, medium size energy plants 100 –1 MW operated by SMEs and central heating plants < 1 MW owned by private entities, public or housing establishments. It is public governance action complementary to the efforts of household owners (see chapter 1.1.1).

1.2.2 Description of measure

To reduce emissions from individual burners it is crucial to support house owners, public entities and entrepreneurs in improving energy efficiency and environmental performance of households. The technical aspects of the measure are described in chapter 1.1.2. This measure comprises the following actions:

National, regional and local energy efficiency programs (retrofitting, replacement of boilers complementary with building thermo insulation and solar panels) and providing economical incentives. Examples in HELCOM countries exist, e.g. supporting house owners in installing solar panels or replacing fuel burners with high efficiency utilities.

Country policies for better quality fuels including biomass, coal, gas and oil (important in Poland). It includes regional contracts between governments and producers and governmental market intervention. The availability of high grade fuels on regional and national scale needs predictable prices as the fuel markets are volatile and political issues have an influence on the market.

1.2.3 Effectiveness

This measure shortens the time of achieving improvement in the residential sector. Current examples show that this instrument is effective as most of the programs are attractive for a wide range of house owners and effective in supporting the investments. For further information on effectiveness of this measure please refer to chapter 1.1.3.

1.2.4 Costs

Costs reflect the availability of technical solutions (see chapter 1.1.4). Using public instruments, the financial burden is split between the public and private sector. In the case of Polish emission reduction programs, the public financial support for house owners (subsidies) varies between 40-80 % of the investment costs and are up to 10 000 Euro. In Germany house owners can get low-interest loans and subsidies up to 13 000 Euro. The costs of running funding programs and public interventions on fuel markets are important. Taking into account investment costs (see chapter 1.1.4), 2 – 5 billion Euro of financial support is needed solely for Poland.

1.2.5 Secondary environmental effects

For secondary environmental effects of this measure please refer to chapter 1.1.5.

1.2.6 Technical feasibility

Several examples for implementation of this measure exist in BSR countries. For technical aspects see chapter 1.1.6.

1.2.7 Secondary socio-economic effects (incl. indirect costs)

The measure is intended to reduce additional costs for households, SMEs and local communities within voluntary participation schemes. Capital and operational future costs can be a problem for consumers. The measure can be adjusted to special socio-economic conditions with support for low income families. For medium size municipal plants e.g. public heating installations, rises in operational costs for the local community can be expected.

1.2.8 Geographical and time scale of effects

Burner replacement is done usually in 20 years period time which is the maximum life span of the utility. The financial support helps in shortening the time of emission reduction to 10 years in comparison with the product life cycle. For further information on geographical and time scale of effects please refer to chapter 1.1.8.

1.2.9 Political enforceability

In general the measure is politically feasible although there are country specific differences. In Poland financial support examples exist but further action is restrained by low budgets on community and regional level. It should be noted that there is no decisive measure tackling the issue of emissions from small and medium energy production installations. A mix of measures including legal, technical, economical and social measures is needed for successful emission reduction, e.g. standardisation of products and financial incentives. Tailored solutions for local needs should be sought.

1.2.10 Cost-effectiveness analysis

Cost-effectiveness for the catchment scale is medium to high (see chapter 1.1.10). Reduction potential is regional and problem specific. This measure is stimulating investments in residential infrastructure. The financial leverage efficiency can be an essential factor in achieving emission reductions.

1.3 Measure 3: Public awareness raising

1.3.1 Description of source

PCDD/F emissions are caused by open burning of agricultural wastes, household waste burning in the open air or in simple installations, field burning, illegal activities in waste handling and recycling (e.g. copper recovery) and other consumer practices like bonfires and grilling. It concerns both, the residential sector and SMEs. On the Baltic Sea catchment scale the reduction potential is up to 10-20% (4% of direct load to the Sea) of the total PCCD/F input to the environment (controlled and uncontrolled sources). This source is especially important in Poland and Russia.

1.3.2 Description of measure

Emission reduction is based on implementation of good practices and good conduct of general public and SMEs. It is applied on national, local and regional scales. This measure comprises awareness raising programs targeting local communities, house owners, individuals, farmers, and small and medium industries including:

- promotion of good practices in maintenance of burners and boilers (HELCOM recommendations),
- discouraging illegal conduct e.g. in collecting and processing of metal scraps, agricultural field burning.

For example, awareness raising programs in Poland and the East Baltic Sea Region (EBSR) are currently being carried out. There are social campaigns aimed to change consumer habits organised on a national level by state institutions. Local initiatives are led by municipalities and non-governmental organisations aimed at awareness raising on environmental risks caused by incineration of wastes in individual boilers and stimulation of bearing the responsibility for the quality of the environment.

1.3.3 Effectiveness

Abandoning the practices like field or trash burning in principle reduce the emission essentially as the concentrations of PCDD/F from burning can reach 100 ng/m^3 in comparison with standard for incineration plants 0.1 ng/m^3. Experiences show that the potential effectiveness of awareness campaigns is rather low in the short term. Since changing the attitudes of people takes a long time, effects can be expected in the long term. For example, despite the efforts undertaken by NGOs, fire fighting services and local administration in Poland in the years 2003–2010, the number of field fires remained at the same level as the practice is seen by farmers as beneficiary for the plant's cultivation. Local continuous activities focusing on target groups, such as young people or women, are especially promising. Sectoral (agriculture, recycling) campaigns are also important. There is however no information how the public react to such campaigns.

1.3.4 Costs

Costs of single and simple actions (leaflets, brochures, posters, internet information) undertaken by local municipalities, local NGOs and other institutions can be relatively low. The cost of a basic information campaign in a medium sized Polish town (100 000 inhabitants) is around 25 000 Euro. Costs for similar country wide campaigns are assumed to be medium.

1.3.5 Secondary environmental effects

No secondary negative effects are expected. Other environmental problems of air emission, waste management and biodiversity are tackled as well (see chapter 1.2.5). This measure is complementary to other measures concerning biodiversity protection or safety issues.

1.3.6 Technical feasibility

Local efficient organisation of waste collection is crucial for this measure. Social and cultural barriers, such as traditions or beliefs, can play a decisive role. There is high potential in all BSR countries of implementation through NGO organisations which are the most suitable and experienced in this activity.

1.3.7 Secondary socio-economic effects (including indirect costs)

No negative socio-economic aspect of this measure is expected. The positive aspect is improvement of air quality and health issues and lowering of fire risks.

1.3.8 Geographical and time scale of effects

This measure is most important in rural and suburban areas, especially for coastline regions. There is a lag between implementation time and consequent achievement of the results of around 10 – 20 years. The most suitable countries are assumed to be Poland, Denmark and Russia.

1.3.9 Political enforceability

In general political enforceability of the measure is high. There is a need for better recognition of the problem by a variety of actors bearing in mind the local traditions and culture. In many cases the problem is not well recognised by the local decision makers. Local scale tions launched by local NGOs and activity groups provide a good stimulus. Cooperation, mutual understanding and support of all stakeholders are crucial. At the same time in all BSR countries various regulations concerning this issue are implemented. Awareness campaigns are usually a part of air emission reduction programs realised by municipalities according to air management requirements[7]. BSR and country wide activity is also recommended to stimulate local communities.

1.3.10 Cost-effectiveness analysis

The cost-effectiveness can be estimated as moderate taking into account relative low costs but also low effects. It should be noted that effects of awareness campaigns are to be expected in future. Social and cultural barriers are very important and influence the effectiveness. From a Baltic Sea perspective this measure is most relevant for rural or suburban areas especially in coastal regions as the dust can be easily deposited directly to the sea or be discharged with storm water. 30% potential reduction of PCDD/F load in Baltic Sea catchment area in residential combustion emissions can be expected in the long term. The cost of simple action can be estimated around 10 –20 million Euro, and with a few years repetition the cost can be doubled with a deposition reduction of up to 100 g I –TEQ.

1.4 Measure 4: Waste management

1.4.1 Description of source

This measure is focused on air emission reduction in the residential sector, agricultural sector and through changes in consumer behaviour including open burning of wastes and other general public activities (see chapter 1.2.1).

1.4.2 Description of measure

Basically the waste management is governed by various EU regulations. Good operational requirements, enforcement and management have an additional emission reduction potential. There is a set of measures concerning common activities of waste management which can be applied appropriately to specific BSR countries situation:

- Waste taxes for establishing obligatory waste management systems to discourage people from waste burning and/or obligatory waste collection with high recovery
- of the materials (e.g. local waste management tax was established in 2011 in Poland).
- Good practices, operational requirements and enforcement (permits) on collection, processing and recycling of wastes.
- Good enforcement of existing regulations on collection, handling and treatment of wastes (permits), management of PCBs and pentachlorophenol containing wastes and boiler ash disposal regulations, waste burning in individual boilers[8], field and backyard burning.

1.4.3 Effectiveness

In principle the reduction potential for emissions caused by open burning of waste is high (around 90 – 100 %). There are already implemented requirements and bans concerning open burning, e.g. as requirement for Common Agricultural Policy (CAP) farmers subsidies. Statistics in Poland show that implementation of this subsidy system associated with the accession to EU was very successful and efficient in cutting the practices. As a result, the number of fires dropped by 50% in 2003 and 2004.

1.4.4 Costs

If well implemented, the measure can be economically efficient but in some aspects the implementation can be difficult to control or would require additional administrative costs. These are incurred in the public sector and depend on existing institutions,

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[7] Air emission reduction programs

[8] There are legal regulations in Poland prohibiting incineration of municipal wastes in boilers which are not adapted to such practices. People who incinerate waste at their homes can be imposed a fine up to 5000 PLN (1265 EUR) or could be imprisoned up to 30 days.

enforcement approaches and practices. The costs of waste collection and disposal are incurred by house owners and other entities.

1.4.5 Secondary environmental effects

Better waste collection incurs additional logistic efforts (energy consumption, emissions) in comparison to in-house burning and local disposal of wastes. For further secondary environmental effects please refer to chapter 1.3.5.

1.4.6 Technical feasibility

This measure requires good institutional arrangements and practices concerning regulation and enforcement. Good legal and administrative culture is the basic precondition.

1.4.7 Secondary socio-economic effects (including indirect costs)

The socio-economic impact is related to costs of proper waste management for households and local communities. Positive economic effects for society should be accounted for, including reduction of fire fighting interventions and material losses due to building fires.

1.4.8 Geographical and time scale of effects

This measure is especially important for rural and suburban areas on the coastline as the hazardous substances are mainly deposited close to the emission source. The time between implementation and the observed results is expected to be around 10 years. The main relevant countries are Poland and probably Russia. Less important are counties like Latvia, Lithuania, Sweden and Germany.

1.4.9 Political enforceability

Legislative changes and establishing implementation and control systems require concerted action on both national and local levels. At the same time, it requires changes in current administrative practices and can have an impact on waste management markets (highly relevant for Poland). Additionally, good public debates with participation of all relevant actors are crucial.

1.4.10 Cost-effectiveness analysis

The cost-effectiveness for the measure is high as the costs for additional improvements in waste collection and disposal are relatively low with the potential of high impacts on emission reduction rates (see 1.1). On the catchment scale, around 20% reduction of the total (100 – 200 g I-TEQ) can be assumed.

1.5 Measure 5: Improvement of BAT and revision of BREF document concerning dust emission reduction

1.5.1 Description of source

The industrial sources comprise combustion sources in energy production, metal industry processes and waste incineration having 16% share in air emissions in BSR countries.

1.5.2 Description of measure

Generally, high reduction potential is expected in smaller installations under 50 MW having high total emissions and for which the regulations so far were less stringent than for Large Combustion Plants (LCP). Emission reduction is required for power plants with a capacity between 1-50 MW with derogations up to 2016 (EC, 2008). New regulation on emission reduction in stationary sources under the Directive on industrial emissions (IED) regime, former Integrated Pollution Prevention and Control (IPPC) regime with capacities > 50 MW has to be implemented in EU countries (EC, 2010). In HELCOM countries by 2008 there were 1257 IPPC installations in energy generation sector and 13262 in other relevant sectors. The existing and proposed EU regulations do not so far cover energy sources with power outputs less than 1MW – this issue is taken into account in chapters concerning the residential sector (chapters: 1.1)[9].

Industrial processes are constantly being improved due to economical and technical changes. The IED (IPPC) Directive requires continuous improvement in sectors and processes which are related to energy transformation, including metallurgy and the energy sector. There is still emission reduction potential in energy generation, waste incineration processes and the metallurgic industry by full implementation of existing BAT standards and by further improvements. In the energy and metallurgical industries, the basic technical option is dust air emission reduction (person contact). New regulations concerning these installations with higher environmental standards consider:

- Large Combustion Plants further dust emissions standards (EC, 2010)
- Air emission standards in 1-50 MW energy plants retrofitted with electrostatic precipitators, fabric filters, and cyclones.
- Air emission standards for 50 KW –1 MW power generation facilities.

Further regulations on emission to air for small and medium waste incineration units, hazardous waste and common waste treatment facilities: sorting, composting and land filling have essential reduction potential.

The measure refers also to voluntary agreements and management systems like ISO 14000 and EMAS securing good practice and conduct.

1.5.3 Effectiveness

The advancement in compliance with BAT approach and potential emission reduction depends on the baseline situation and site specific conditions of any particular facility. The improvement has the biggest potential in installations with no existing abatement measures or low efficiency such as multicyclones (50% efficiency). Additional possible reduction is as follows: for 5-50 MW energy installations using 1-stage Electrostatic Precipitators (ESP) gives 93% reduction, using fabric filter: 99.7%, and applying 2-stage ESP: 96 %; and for large power plants, a combination of 2- 3 stages ESP + wet Flue Gas Desulphurisation (FGD) achieving 99% efficiency (Karvosenoja, 2007). For medium installations in Poland the reduction potential is high (up to 50%) according to current regulations with the target level in 2016.

1.5.4 Costs

Additional costs for advanced abatement techniques in comparison to current cost of air abatement technologies are assumed to range between low to moderate. These are as follows: 380-480 Euro per Mg of fine dust emissions (PM 2.5), 2- 3 stages Electrostatic Precipitators (ESP) + wet Flue Gas Desulfurization (FGD), 1-stage ESP 5-50 MW solid fuel 260-2300 Euro per Mg of dust (g of I-TEQ), for <50 MW 220-13000 Euro per Mg. Fabric filter installation costs were reported in the range of 330 to 2900 Euro per Mg of dust. For industrial processes ESP/ESP + scrubber/fabric filter abatement costs is between 17-1500 Euro per Mg of dust (mg of I-TEQ) (Karvosenoja, 2007).

1.5.5 Secondary environmental effects

Dust removal in wet systems generates additional waste and effluents contaminated with PCDD/F which have to be treated in appropriate facilities. The operation of electrostatic filters and treatment facilities also cause an additional energy demand. Improvement in the combustion process leads to a reduction in fine dust emissions

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[9] The problem of energy waste management should be also taken into account as well as management schemes for hazardous waste management in industry, landfill effluents quality standards and appropriate treatment regulations

(PM2.5 and PM10) and other contaminants including heavy metals and Polybrominated Diphenylethers (PBDEs).

1.5.6 Technical feasibility

Improved technologies for air emission abatement exist in several Baltic Sea countries including basic dust reduction measures (electrostatic precipitators, fabric filters, wet scrubbers) and advanced measures which are process specific air abatement techniques. For installations with capacities of 50 kW to 50 MW, improved simple and advanced electrostatic precipitators, fabric filters and cyclones can be recruited. The particular application depends on specific type of industrial process and type of fuel. For example for solid fuel burning power plants and industrial facilities of 5-600 MW capacity, emission reduction is through installation of fabric filters. For heavy oil-fired boilers with 1-5 MW capacity, electrostatic precipitators (ESP) are used and for coal fired power plants with capacities of 300 to 1300 MW, 2- 3 stages ESP+ wet Flue Gas Desulphurisation (FGD) systems are used (Karvosenoja, 2007). For PCDD/F removal, activated carbon techniques and afterburners are applied (BREF).

1.5.7 Secondary socio-economic effects (including indirect costs)

The additional costs for enterprises can have some negative effect on their economical performance. Energy prices for the consumers and industries can be raised. In general the socio-economical impact is low.

1.5.8 Geographical and time scale of effects

This measure is the most relevant for Poland, since the share of industrial emissions is 11%. Concerning other Baltic Sea countries please refer to different shares, presented ion the Guidance Document on Dioxins). The time for implementation is relatively short (5-10 years) depending on legal specifications.

1.5.9 Political enforceability

Political enforceability varies between moderate to high. The emission standards and Best Available Technique (BAT) approach is decided on EU level along with other policies including revision of BREFs, for energy and metallurgic sectors and PM 2.5 oriented policies. Implementation of new air emission standards (ELV) for further reduction in 1-50 MW energy plants retrofitted with electrostatic precipitators, fabric filters and cyclones. EU regulations governing actions concerning particulate matter reduction results also in PCDD/PCDF emission reduction (EMEP 2009).

The enforcement can be strengthened by good implementation practice with relevant consultancies and administrative procedures with monitoring, treatment requirements and emission levels to environment being well defined.

1.5.10 Cost-effectiveness analysis

Cost-effectiveness is assumed to be low to medium for Large Combustion Plants (LCP) in comparison to the existing situation as most of the facilities in Baltic Sea countries already meet high standards of particulate matter reduction. Gradual improvement along with further changes in BAT practice is the best approach with more cost-effective solutions applied.

The theoretical potential for reduction of PCDD/F load to Baltic Sea through air deposition exists in large industries such as metal production, energy sectors and waste management. In medium enterprises reduction potential is up to 15 – 20 % (controlled sources). For the whole catchment area the share is lower around 10%. According to a Finish study, industrial-scale combustion using solid fuel boilers and fabric filter technologies is assumed to be relatively cost-efficient. The emission of 1.2 Gg per year could be reduced with marginal costs below 5000 € per Mg of dust in boilers larger than 5 thermal megawatts (MWth). Less cost-efficient reductions of 1.8 Gg per year are achievable in smaller solid fuel and heavy fuel oil boilers by Electrostatic Precipitators Technology (Karvosenoja, 2007).

1.6 Measure 6: Treatment of contaminated soil

1.6.1 Description of source

There are two potential main sources of waste contaminated with PCCD/F: mineral oil with PCB impurities and wood with pentachlorophenol wastes contaminated with PCDD/F. These wastes are relevant for all Baltic countries especially Sweden, Finland and Poland. Any sites contaminated with those materials that are located within the drainage area of the Baltic Sea are potential sources of PCDD/F. These include old landfills, illegal dump sites and other residual contaminations. The emissions are also relevant at industrial sites due to PCB and pentachlorphenols production or use. Only a few cases of PCDD/F contamination in the drainage area of the Baltic Sea were researched. It is estimated that in 2001, there were approximately one hundred paper mills in the Baltic Sea drainage area, with three of them using elemental chlorine for bleaching. These three paper mills were all located in Kaliningrad, Russia (Leithe-Eriksen, 2001). In addition to that, there exist former production sites, called "Organika-Rokita" in the Polish town of Brzeg Dolny. Moreover, the use of the commercial product “Ky 5” in Finland has led to soil contamination by chlorophenols and PCDD/Fs at sawmills and waste disposal sites in the Baltic Sea region (Weber et al, 2008).

1.6.2 Description of measure

Soil and sediment contaminations have to be treated on a case by case basis, appropriate to site conditions and contamination levels. After careful consideration, the treatment of contaminated soils can be considered only in a few specific cases. For the treatment of sediments the two technical measures dredging and capping are specified. For contaminated soils a variety of approaches exist.

1.6.3 Effectiveness

It is expected that removing and/or capping of contaminated sediments between the source can significantly reduce PCDD/F downstream transport preventing flush of the sediments during high flow events. In a Finish case from 36,000 m^3 contaminated sediments approximately 40% is estimated to be flushed away during a period of 30 years (Weber et al. 2009). Also land contamination can be of importance due to the potential of erosion or run off but this is very site specific.

1.6.4 Costs

In the mentioned Finnish case the anticipated restoration costs for the whole Kyumoki river trough sediment dredging are estimated to be approximately 480 million Euros. In general, cost-efficient site specific approaches should be applied to reduce costs.

1.6.5 Secondary environmental effects

Dredging of contaminated soil can lead to temporary increase of PCDD/F concentrations in downstream waters and surface sediments. In the worst case scenario, a 10% resuspension of the dredged sediment can occur but usually resuspension rates range between one and two percent. Additionally, other pollutants such as mercury can be resuspended.

1.6.6 Technical feasibility

The techniques are well known but require careful site specific approaches. Only a few Finnish sawmill sites contaminated with chlorophenols and PCDD/Fs have been remediated. For highly contaminated sediments in hot spot areas, technical measures can be considered but in general the most important measure is monitoring and risk control.

1.6.7 Secondary socio-economic effects (including indirect costs)

For local communities living in highly contaminated areas very positive effects are expected.

1.6.8 Geographical and time scale of effects

The problem is relevant for all Baltic Sea countries especially for Finland, Sweden and Poland.

1.6.9 Political enforceability

In Sweden, Germany and Finland a systemic approach to contaminated land remediation exists.

1.6.10 Cost-effectiveness analysis

Cost-effectiveness can be high in particular localities, as in the Finnish cases. Dredging at site, treatment and a close disposal of the most contaminated sediments in Kyumoki river case (90,000 m^3) near the pollution source showed the best cost-effectiveness: The anticipated costs range from 5.5 to 8.3 million Euros (Verta, 2009). In general, proper dredging is potentially the most cost-effective measure.

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II. Organotin compounds: Tributyltin compounds (TBT) and Triphenyltin compounds (TPhT)

2.1 Measure 1: Sediment management – environmental dredging

2.1.1 Description of source

TBT/TPhT binds easily to particulate matter and contaminates the sediments (Feenestra et al. 2009). The half-life is considered to be between 1-2 years in aerobic sediments and up to ten times longer in anaerobic sediments. After many years of TBT use as a biocide in antifouling paints, the sediments, especially in shipyards and harbours, contain high amounts of TBT. A resuspension (from 0.1 to 5%) of such contaminated sediments may release a high concentration of organotins into the water during dredging of such sediments. As most of the dredging takes place for navigational purposes, the concentrations of TBT in sediments in harbours and shipyards were taken for further calculations, i.e. the range of median concentrations 370 – 7535 μg/kg dry weight. The amount of dredged materials in the BSR is 10 million tonnes (dry weight) or 7million m^3 (amount of sediments dredged and dumped in the sea, see description of B2). The overall amount of TBT with dredged materials is 3.7 – 75.35 t of TBT.

2.1.2 Description of measure

Environmental dredging is a safe and precise removal of contaminated sediment. Environmental dredging minimizes the resuspension of contaminants, enables the complete remove of contaminants, minimizes the amount of water removed and reduces over dredging (excessive removal of clean sediment). Two basic types of dredgers can be used: mechanical and hydraulic (or hybrid). For better environmental performance, the mechanical dredgers have overlapping sides, rubber seals and a special venting system. A cutter-head of hydraulic dredge is shielded to control resuspension and minimize inflow of water. Both types of dredgers are equipped with additional instruments, such as monitors (video, sonar, etc.) to provide feedback to operators and Global Positioning Systems (GPS) for improved dredging precision. Work areas can also incorporate multi-station real-time monitoring as an early warning system to help gauge for any significant re-suspension of sediment or spillage in the water during dredging activities. Solid barriers, like coffer dams, sheet pilings, silt curtains and silt screens are placed around dredge sites to keep resuspended sediments from flowing downstream.

2.1.3 Effectiveness

The effectiveness of method, i.e. reduction of TBT concentration can vary from 50 to 100% depending on the chosen method for dredging (Socopse project).

2.1.4 Costs

Costs are depending on site and sediment conditions, treatment technique etc. While performing bigger jobs the cost per cubic meter becomes cheaper because of economies of scale. Since cost differences between different types of dredges are small, costs should be one of the last considerations when choosing a dredge. The physical site constraints and the requirements for treatment and disposal of the dredged sediment are of primary importance. Costs are expected to be approximately €13 to €44 per cubic meter.

2.1.5 Secondary environmental effects

The environmental dredging affects all pollutants, which tend to be bound on to sediments, e.g. heavy metals, phthalates, PCB etc. It also reduces excessive removal of clean sediments and water. Furthermore it much less disturbs the surrounding environment for the living organisms and is less harmful to aquatic fauna concerning turbidity of the water body.

2.1.6 *Technical feasibility*

Environmental dredging is a proven technology, which could be widely used without specific limitations or high complexity of implementation. The choice of technique is more dependent on site and sediment conditions.

2.1.7 *Secondary socio-economic effects (including indirect costs)*

As in general, there are no studies that could be referenced regarding secondary socio-economic effect but case by case analysis allow us to maintain that indirect costs can vary from 15 % to 30 % of the total cost.

2.1.8 *Geographical and time scale of effects*

The measure becomes effective immediately while applied. The measure can have mostly effects on a watershed level as dredging can affect a large surrounding area.

2.1.9 *Political enforceability*

Environmental dredging approaches and some techniques are already recommended to BSR countries by HELCOM recommendation 13/1 Disposal of dredged spoils. There are no special laws in the EU for dredged material, but a number of EU Directives have a direct or indirect impact on the management of dredged material. The impact of the Water Framework Directive is likely to have mainly indirect effects as a result of complicated project development. Direct effects may result from additional monitoring requirements during and after completion of projects as imposed by the respective permitting authority in member states (Terra et Aqua | Number 104 | September 2006).

2.1.10 *Cost-effectiveness analysis*

Based on the data presented in the proceeding chapter, two scenarios for cost-effectiveness analysis (CEA) are derived.

The worst case scenario for CE (low concentration – low resuspension – low reduction rate – high cost)

CE: €154 million /1.85 kg = **€83 million /kg of TBT.**

The best case scenario for CE (high concentration – high resuspension – high reduction rate – low costs)

CE: € 46.2 million / 3770 kg = **€12 250 / kg of TBT**.

More details on calculation in Annex B1 of the Guidance Document on Organotins (TBT, TPhT) .

2.2 Measure 2: Sediment management – no dumping of TBT/TPhT sediments in the Baltic Sea

2.2.1 *Description of source*

As reported to HELCOM, each year BSR countries dump approximately 10 million tonnes (dry weight) of dredged materials to the Baltic Sea. For example Finland has measured the concentration of TBT in sediments of dumping site dredged materials. As a result, concentrations varied from <1 to 580 μg/kg as TBT (dry weight), which is much higher than the predicted no effect concentration (PNEC) for benthic organisms. Most countries control heavy metals in such dredged materials but only a few cover organotins as well. As reported by three countries, average concentrations of TBT/TPhT in dredged materials (mg TBT/TPhT/t dredged material) varied from 1.2 to 47 mg TBT/t and 0.06 mg TPhT/t. The total possible load to the whole Baltic Sea within dredged materials could be estimated as 12 - 470 kg of TBT and 0.6 kg TPhT per year.

2.2.2 *Description of measure*

As TBT is considered as a highly toxic pollutant (the PNEC fro benthic community and EQS for surface water are very low), the option of no dumping of sediments with

TBT/TPhT is considered. In this case other methods must be applied, such as solidification/ stabilisation of dredged materials.

2.2.3 Effectiveness

The reduction potential of TBT/TPHT through this measure is 100%.

2.2.4 Costs

As the dredging of sediments for navigational purposes should take place anyway, the costs of this measure were looked at from the perspective of additional costs raised by alternative methods applied for utilisation of dredged materials. Costs for these alternatives can be very high. Instead of dumping sediments into the sea the alternatives looked at are land filling, which seems to be costly (150-200 €/m^3; Svedberg Bo et al., 2010). A more cost-effective and sustainable measure compared to deposition at land or at sea are stabilisation or solidification (20-50 €/m^3, Svedberg Bo et al., 2010). These costs exclude common costs for dredging, transportation etc.

2.2.5 Secondary environmental effects

No dumping of dredged sediments to the sea also would highly reduce input of other pollutants bound to the sediments (e.g. heavy metals). However, if not properly controlled, alternatives could have negative impacts to the environment (e.g. intensive land use for landfilling, emissions to air for transportation and leachate from landfills).

2.2.6 Technical feasibility

There are technically proven technologies available.

2.2.7 Secondary socio-economic effects (including indirect costs)

Solidification or stabilisation of dredged materials is considered to be cost-effective and sustainable compared to deposition at land or at sea. Using the technique could also enable a beneficial use of stabilised sediments for land reclamation at exploration for port facilities, industrial activities or other use. In addition, the need of natural resources and related transportation could decrease environmental advantages. The waste from other sectors (e.g. fly ash) could be used as a binder instead of landfilling it. Landfilling is a land use intensive measure that could induce additional costs for citizens.

2.2.8 Geographical and time scale effects

The development of proper facilities from landfills or case-specific investigations and tests for use of dredged materials for solidification or stabilisation would take from one to several years. Therefore it could be targeted to more contaminated sediments first. For that BSR countries must set the limits also for organotins in the sediments that could be disposed to the Baltic Sea.

2.2.9 Political enforceability

Up to now a common legislation on the management of dredged material has not yet been proclaimed at European level, though the concept at the basis of each European country regulation is to consider sediments as a resource and not as waste. For example in Finland if TBT does not exceed limits of 0.15 µg /l in interstitial water, it can be disposed of at a an approved open sea disposal site ((CE 83/2001 (DS) Peng Chau Sewage Treatment Works Upgrade – Investigation, Design and Construction Environmental Impact Assessment Report (Final)).

2.2.10 Cost-Effectiveness Analysis

Based on the data presented in the proceeding chapter, two scenarios for cost-effectiveness analysis (CEA) are derived.

Table 5: Cost effectiveness analysis for no dumping of sediments into the sea

	Min load of TBT, kg	Max load of TBT, kg	Load of TPhT, kg
	12	470	0.6
Costs for elimination of 1 kg of pollutant by landfilling of dredged sediments	€83-116 million /kg	€2-3 million /kg	€1.6-2.3 billion /kg
Costs for elimination of 1 kg of pollutant by solidification/stabilisation of dredged sediments	€11-29 million /kg	€0.3-0.7 million /kg	€0.23-0.58 billion /kg

The worst case scenario for CE (low concentration – high cost of alternative)

CE: €116 million /kg of TBT

CE: €2.3 billion /kg of TPhT

The best case scenario for CE (high concentration – low cost of alternative)

CE: €0.3 million / kg of TBT

CE: €0.23 milliard /kg of TPhT

More details on calculation in Annex B2 of the Guidance Document on Organotins (TBT, TPhT).

2.3 Measure 3: Sediment management - ex situ solidification and stabilisation

2.3.1 Description of source

There are many “hot-spots” with highly contaminated sediments in coastal areas, ports, estuaries etc. being an important secondary source of TBT/TPhT.

2.3.2 Description of measure

As for solidification/stabilisation (s/s) contaminants are physically bound or enclosed within a stabilised mass (solidification), or chemical reactions are induced between the stabilising agent and contaminants to reduce their mobility (stabilisation). Ex situ s/s treated masses may be used as construction material (if human and environmental impacts from the treated contaminated masses are reduced enough) or requires disposal of the resultant materials.

2.3.3 Effectiveness

Applicability and effectiveness of the method is dependent on contaminants and site characteristics. The target contaminant group for ex situ s/s is mainly inorganics, including radionuclides. Most s/s technologies have limited effectiveness against organics and pesticides, except vitrification, which destroys most organic contaminants. However the Pozzolan/Portland Cement method, in which new innovative forms of binders/mixtures (other than, or in combination with, cement) with/without additives have been tested in pilot and full scale in the last years, has shown significantly reduced leachability of organics from stabilised contaminated sediment, compared to the contaminated sediment itself. Case studies showed that also significantly reduced leachability of TBT could be achieved with this method.

2.3.4 Costs

Costs vary from €100 - €250 per m^3 (Larsson et al., 2011, SMOCS) to €20 - 50 per m^3 (Svedberg Bo et al. 2010).

2.3.5 Secondary environmental effects

The measure also targets other pollutants, such as heavy metals, polycyclic aromatic hydrocarbons (PAHs), polychlorinated biphenyls (PCBs) and radionuclides. Positive environmental impacts are the reduction of permeability and leachability of contaminants compared to original matrix. But raised pH might have local negative impact. The product could also be used as a construction material replacing the use of other natural resources. Industrial by-products or wastes, such as fly ashes, slags, gypsum products and limes can be utilised in binder admixtures, which decreases the need for commercially available binders and saves binder costs.

2.3.6 Technical feasibility

The method is fully developed, applied and technically reliable. However, thoroughly designed laboratory tests are necessary to evaluate the site specific suitability of method. There are some limitations as listed below:

- Environmental conditions may (theoretically) affect the long-term immobilisation of contaminants.
- Some processes result in a significant increase in volume (up to double the original volume).
- Certain wastes are incompatible with different processes. Treatability studies are generally required.
- High to very high organic content (more / much more than 10 w-%) may reduce the cementing/hardening process.
- Long-term effectiveness (> ca. 30 years) has not yet been demonstrated for many contaminant/process combinations.
- The method will not lower leachability of all contaminants.

2.3.7 Secondary socio-economic effects (including indirect costs)

The product could be used as construction material replacing other natural resources for quay, building foundations, and base for parking lots, capping or containment of contaminated solid/sediments at deposits. No land for landfills is needed. Industrial by-products or wastes such as fly ashes, slags, gypsum products, and limes could be utilised avoiding landfilling and reducing costs.

2.3.8 Geographical and time scale effects

As the content of dredged materials varies considerably, each time an investigative testing phase should be undertaken before the direct application, i.e. technical tests, different compositions of binders and leachability of contaminants.

2.3.9 Political enforceability

Dredging operations are carried out in full compliance with relevant legislation. It is important that during further development of the legislative framework the necessity of dredging and the usefulness of dredged material is recognised. EU member states need to comply with a wide variety of EU environmental regulations in the area of water quality and waste management that have a bearing on dredged material transport, use and disposal. As these provisions have rarely been designed particularly for dredged material, member states continuously face interpretation questions – the answers to which may vary from one occasion to the other - having high potential impact on relocation, transport or disposal options (www.bafg.de).

2.3.10 Cost-Effectiveness Analysis

Based on the data presented in the proceeding chapter, two scenarios for cost-effectiveness analysis (CEA) are derived.

Worst case CE scenario for TBT: low load – high price

CE = 7million m^3*50 €/m^3 /12 kg = €29 million /kg TBT

Best case CE scenario for TBT:

CE = 7million m^3*20 €/m^3 /470 kg = €0.3 million /kg TBT

More details on calculation are found in Annex B3 of the Guidance Document on Organotins (TBT, TPhT).

2.4 Measure 4: Sludge treatment – controlled incineration

2.4.1 *Description of source*

Butyltin compounds like TBT, DBT and MBT are widespread organotin compounds at MWWTP. The highest concentrations in the wastewater treatment plants were found rather in inflow wastewater and sludge than in effluents. This shows that WWTPs generally exhibit efficient removal of organotin compounds from wastewater into sludge. TPhT was not found in wastewater (not even in untreated). The concentrations of TBT in sludge varied from 0.04 to 98 μg/kg dry weight (Mehtonen J., 2009). Total production of sludge in BSR is 1.263 million tonnes and land application (agricultural use and composting) is 0.2899 million tonnes. TBT input to the land in BSR through application of contaminated sludge is 24 g to 57 kg. By reusing sewage sludge as a fertiliser in agriculture, organotins again are released to the environment. TBT can leach to groundwater, from where it can be introduced into the food chain.

2.4.2 *Description of measure*

No land application of sludge contaminated with TBT/TPhT would eliminate the input of organotins to the environment. However, in this case other utilisation methods must be applied for utilisation of sludge. Therefore, landfilling and incineration of sludge can be considered as alternatives.

2.4.3 *Effectiveness*

Incineration of sludge contaminated with TBT/TPhT would eliminate the input of organotins to the environment significantly.

2.4.4 *Costs*

Costs of this method would be equal to the costs raised by applying alternative methods for utilisation of sludge, e.g. incineration or landfilling.

Table 6: Sludge disposal costs, €/t of dry weight

Measure	EC, 2002	RPA, 2008
Agriculture	110- 160	126-185
Composting	210-250	280
Incineration	260-350	339-417
Landfilling	260-350	309

2.4.5 *Secondary environmental effects*

The sludge from WWTPs contains a variety of other pollutants, especially those which tend to bind to particles, e.g. heavy metals, alkylphenols, phthalates etc., which also re-enter the environment upon application of contaminated sludge to the land. They may be transferred to soils, leach to groundwater, volatilise to air or be uptaken by plants and in such way introduced into the food chain. Alternative methods may also have negative impacts on the environment. For example, incineration may generate emissions to the air (particles, acid gases, greenhouse gases, heavy metals, volatile organic compounds, etc.), to soil (disposal of ashes and flue gas treatment residues, atmospheric deposition) and water (wet flue gas treatment processes). Landfill operation can generates emissions to the air (e.g. greenhouse gases like methane and carbon dioxide), and to soil and to water (e.g. heavy metals, organic compounds and micro-organisms).

2.4.6 Technical feasibility

Both alternative technologies are well-developed and applied on a full scale.

2.4.7 Secondary socio-economic effects (including indirect costs)

Land spreading of sludge or sludge-derived material partially replaces the use of conventional fertilisers. Both the incineration of sludge and biogas which is generated in landfills can be used to produce energy. However the alternatives would still raise the overall cost for wastewater treatment which would affect citizens connected to WWTPs.

2.4.8 Geographical and time scale of effects

New sludge incineration facilities must be built in eastern BSR countries, since in Estonia, Latvia and Lithuania no incinerators exist. In Lithuania municipal landfills do not accept sludge. Therefore it is temporarily kept in storage fields. Thus it would take several years to implement these measures. Incineration of sludge is widely used in Denmark and Germany and in several Russian cities some big plants are currently being constructed.

2.4.9 Political enforceability

The requirements for sludge application on land include limits for heavy metals but not for organics, such as TBT/TPhT. Germany has mono-incineration when sludge is incinerated in dedicated incineration plants.

2.4.10 Cost-Effectiveness Analysis

Based on data presented in the proceeding chapter, two scenarios for cost-effectiveness analysis (CEA) are derived. For the calculation of cost-effectiveness (CE) the incineration and landfilling were taken as alternative methods and therefore only differences to the costs for sludge on land were considered.

Table 7: Costs

Total TBT load for sludge to land	Amount of sludge applied to land (agriculture+compost)	Additional costs for sludge disposal compared to other methods (landfilling, incineration)	TBT reduction potential
0.024-57 kg	581 300 tonnes	€124-291 /t dry weight	100%

Worst case scenario: low load-high costs

CE: 581300*291/0.024 = €7 billion /kg TBT

Best case scenario: high load-low costs

CE: 581300*124/57 = €1.3 million /kg TBT

2.5 Measure 5: Waste management - controlled land filling

2.5.1 Description of source

TBT/TPhT contaminated dredged material can possibly be disposed of in controlled landfill sites. TBT can be contained within organotin waste which is generated through the removal of anti-fouling paint containing organotin chemicals.

2.5.2 Description of measure

According to legal requirements, organotin waste should be disposed in controlled landfills. It is specified that TBT contaminated sediment must be disposed in separated mono cells and controlled through a leachate barrier and leachate collection system. An application for the variation of landfill licence and a supporting design documentation is required.

2.5.3 Effectiveness

Controlled landfilling of sludge contaminated with TBT/TPhT would eliminate the input of organotins to the environment from this source by 90%.

2.5.4 Costs

The use of landfill has always been the cheapest waste disposal option, but cost of landfill as a waste disposal option is rising rapidly. The situation as well as the cost incurred by handling of hazardous waste varies from landfill to landfill and according to excavation volumes. Landfilling costs vary from €150-200 /m^3 (Svedberg Bo et al., 2010) over €260-350 /t (EC, 2002) to €309 /t (RPA, 2008).

2.5.5 Secondary environmental effects

Possible secondary environmental effects are rainwater penetration into the mono cell causing migration of sediment containing TBT into the landfill.

2.5.6 Technical feasibility

TBT containing sludge after primary and secondary sedimentation treatment must not be dumped at sea or re-used as a fertiliser. Landfilling (after pretreatment) or incineration are preferable options. (SOCOPSE)

2.5.7 Secondary socio-economic effects (including indirect costs)

Lack of landfill space increases costs. During postclosure all leak detection systems and groundwater monitoring systems are required to be maintained. Postclosure care usually lasts 30 years after closure.

2.5.8 Geographical and time scale of effects

Landfilling is not preferred in all BS countries due to a lack of land.

2.5.9 Political enforceability

As TBT and TPhT derive from common consumables, the absence of such substances in waste can be achieved only by total ban of use in manufactured goods.

2.5.10 Cost-Effectiveness Analysis

Worst case scenario: low load-high costs

CE: €116 million /kg TBT

Best case scenario: high load-low costs

CE: €2 million /kg TBT

2.6 <u>Measure 6</u>: Wastewater treatment in shipyards

2.6.1 Description of source

It is estimated that between 70% and 80% of the global merchant fleet has tributyltin (TBT) antifouling paint applied to their hulls. When these ships are dry-docked, shipyards can generate tens of thousands of gallons of TBT contaminated washwater.

2.6.2 Description of measure

Different types of particular waste must be stored separately while awaiting treatment and disposal. Contaminated wastewater should be left standing so that suspended particles can settle and be physically separated from the water. The permits may be granted for treating TBT contaminated wastewater.

There are three main methods used for the removal of anti-fouling systems – scraping, blasting and water blasting/washing. Work should be carried out in enclosed areas or in a dry dock with features allowing the collection of liquids containing contaminants.

One way for removal of organotin compounds is flocculation adsorption. The powdered activated carbons exhibit a large organotin adsorption ability.

2.6.3 Effectiveness

Coagulation/flocculation is a suitable technique for the removal of suspended solids and colloidal material, to which micro-contaminants like TBT are attached. Both ferric sulphate and aluminium sulphate (alum) can be used for TBT removal from shipyard waste water. In a full scale treatment plant study a removal of 99,8 % of TBT was achieved with a treatment system consisting of coagulation-clarification, filtration and activated carbon adsorption. Removal values of 87% for TBT were found for the coagulation step in laboratory experiments.

2.6.4 Costs

For heavily contaminated waters (50 µg Sn/l) the operating costs exceed €4/m^3 for an organotin discharge limit of 0.1 µg Sn/l, which is not economically acceptable. For a discharge limit of 0.5 µg Sn/l the total operating cost is about €2.5/m^3. Increasing the discharge limit beyond this value does not result in a large decrease in operating cost. An organotin discharge limit of 0.5 µg Sn/l is thus economically and technically the best option.

2.6.5 Secondary environmental effects

Disadvantages are the additional use of chemicals and the subsequent chemical sludge production. The dosed coagulant and flocculent emerge in the residual waste stream as a chemical sludge.

2.6.6 Technical feasibility

In many shipyards in BSR countries ships are treated in open docs and the wastewater goes to the surface water through simple mechanical filters. Changing filtration systems in shipyards which receive approximately 120-250 boats per year, might require investments of €13000-25000 (BalticSeaBreeze).

Investment costs are relatively low, except for the coal bed itself. Operational costs are very strong depending on the type of coal that is used and the dosing rate of the coal (range €0.1 to 1.0 /m^3). Typical coal costs are €1 to 5/kg. During dry sandblasting approximately 400 m^3 of wastewater is generated from large ships, while during hydro blasting 1600 m^3 of wastewater is generated.

2.6.7 Secondary socio-economic effects (including indirect costs)

Used adsorbents should be refined, causing additional costs.

2.6.8 Geographical and time scale of effects

Implementation is possible in all BSR countries.

2.6.9 Political enforceability

Many countries still lack regulation for dockyard discharges.

2.6.10 Cost-Effectiveness Analysis

Worst case scenario: low load-high costs

CE: €5 million /kg TBT

Best case scenario: high load-low costs

CE: €80 000 /kg TBT

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III. Polybrominated Diphenyl Ethers (PBDE)

3.1 Measure 1: Sludge treatment – controlled incineration

3.1.1 Description of source

As the majority of PBDE emissions come from diffuse sources, such as consumer articles during their service life, and since PBDEs due to their physical and chemical properties bind to solid fraction in the aquatic environment, it is very important to correctly manage the matrix where most of these flame retardants accumulate – sludge of municipal WWTPs.

3.1.2 Description of measure

The options for dealing with sludge vary from depositing it in landfills, applying it to agricultural and forest soils as a fertilizer, to incineration in hazardous waste incinerators. Incineration is usually done in specially designated facilities, which either can be dedicated only to sludge incineration in cases when incoming loads are quantitatively large or can incinerate other types of hazardous waste as well. The most common technologies used for sludge incineration are hearth incineration and fluidized bed incineration. Each system has its advantages. Hearth incinerators are the most widely used at the moment – movement of sludge through hearths exposes a larger area for incineration than other technologies, making it very efficient. Fluidized bed incinerators, however, allow the utilization of energy generated during the incineration process, thus leading to cost savings. In general, the concrete technology should be chosen depending on site-specific and local conditions.

3.1.3 Effectiveness

In terms of reducing potential emissions, incineration is one of the most effective options, as it eliminates virtually all traces of most organic pollutants in the sludge, including PBDEs, preventing any further emissions.

3.1.4 Costs

Incineration can be a very costly option, especially if no local/regional incinerators are available. At the same time proper application to farmland can also be expensive, as it requires regular monitoring/taking of sludge samples and analysing them not only for traditionally monitored heavy metals but also for organic pollutants.

As a result, the costs and suitability of each solution can vary considerably depending on local and regional circumstances.

Generally speaking, the highest costs are investments to build an incineration facility. Operating costs can vary depending on the amounts of sludge incinerated, sludge moisture content, whether other types of waste are co-incinerated, etc. In the long term investments might pay back and become more feasible, especially if the incinerator can generate heat or electricity which can be sold to consumers and if sludge would otherwise have to be transported over long distances or exported for utilization.

A general assessment of the costs of sludge management options was made in a research project on behalf of the European Commission. Although this research was done in 2002 and absolute numbers might have changed, the relative proportion of costs between different management options should be approximately the same.

As can be seen, in terms of direct costs incineration is the most expensive option, both due to investments needed and costs of regular maintenance, summing up to just over 300 Euros per ton of dry mass (tDM).

3.1.5 Secondary environmental effects

Secondary environmental effects can include increased air pollution levels in areas around incinerators, however, modern air purification techniques achieve very high removal efficiencies, so that what is emitted to air is practically only pure water steam.

Nevertheless, flue gas emissions from incineration processes must be monitored regularly.

3.1.6 Technical feasibility

There are no known technical difficulties to implement this measure practically anywhere, if finances allow it. Several specialized companies across the world are offering to build such installations, employing different incineration techniques and at differing costs.

3.1.7 Secondary socio-economic effects (including indirect costs)

The secondary socio-economic effects are generally positive, as building and operating such installations creates work places, provides local municipalities with heat and electricity, etc. Unwanted effects might arise if funds for building such installations are raised by increasing fees for public services, such as collection of waste water. Financing possibilities must be investigated beforehand and need to be carefully evaluated, taking into account that the costs will probably amount to several million EUR.

3.1.8 Geographical and time scale of effects

The effects of implementing this measure should be clearly noticeable as soon as sludge is no longer mismanaged. It might take some time for particular environmental compartments to clear up from previous pollution, but this should not take longer than a couple of years.

3.1.9 Political enforceability

This measure should be well aligned with other environmental goals, for example, reducing amounts of waste, especially hazardous waste. As sludge is produced in rather high quantities, managing it can become a handful if there is no concrete plan for what to do with it, as is currently the case in the Baltic States, for example.

3.1.10 Cost-effectiveness analysis

Cost effectiveness depends mainly on three factors: how much sludge is to be incinerated annually, are there other types of waste incinerated as well and whether the excess heat can be sold in the form of electricity or heating.

As already described in section 6.1.4., direct costs for incineration, especially forsludge mono-incineration, are higher than those of other types of sludge management. However, for cost calculation, side costs must also be taken into account – especially the costs connected to controlling sludge quality and determining whether it is sufficient for application to agricultural soils. Here analytical and monitoring costs come into play which can significantly increase total management costs per ton of sludge. Although total costs for other means of management besides incineration would depend greatly on numerous factors, like availability of laboratory capacity and enforcement level of sludge quality testing, it is quite likely that regular costs of incineration would in the long term be lower than those of agricultural use with regular quality checks for contents of organic pollutants, especially if co-incineration of other waste types and/or heat and electricity production can be done.

3.2 Measure 2: Substitution of PBDEs in polymers, textiles and construction materials

3.2.1 Description of source

As described before, a large part of emissions are generated by use of ready products, mostly consumer articles. Dealing with such types of pollution by end-of-pipe measures can be very challenging because it is almost impossible to acknowledge all the possible places where it might end up. Substitution of pollutant at source solves this issue.

3.2.2 Description of measure

Several US and EU studies over the last decade have shown that there are sufficient and affordable alternatives among non-halogenated flame retardants for all most commonly used PBDE congeners and for all industries and products, especially for the ones where PBDEs are most commonly used – consumer electronics (TV sets, monitors, cell phones) and textiles.

3.2.3 Effectiveness

Substitution can be evaluated as the most effective measure to reduce total emissions – by using flame retardants other than PBDEs, the very source of emissions is ultimately eliminated.

3.2.4 Costs

The table below compares the most commonly used alternatives, their median prices and possible affordability issues for producers.

PBDEs used in...	Possible substitutes	Price of substitutes (EUR/kg)	Affordability issues for producers
HIPS/PPO; PC/ABS	Resorcinol bis-diphenylphosphate (RDP) - cannot be direct substitute for pure HIPS	<3.5	Minimal or none
	Triphenylphosphate (TPP)	6	
	Bisphenol A	4.5	
Textiles and upholstery for furniture	Phosporus-based flame retardants, such as Pyromescent	5.5	Minimal, except for transportation applications where they can be significant
	Mixed phosphorus/halogenated flame retardants, such as Pyrozoyl 6P	6.5	
Building and construction materials	Different types of phosphorus-based products, such as Reofos and Kronitex	4	Minimal or none

When evaluating the prices for alternatives, it must be taken into account that the average decaBDE market price is 2.5 – 3 EUR/kg. The price for pentaBDE was 6 EUR/kg in 2005 when it was phased out. Current prices vary according to supplier, but as penta- and octa- congeners are banned for import and production in all countries around the Baltic Sea, comparing current prices does not seem to be a relevant issue.

The total cost increase for ready products is rather insignificant: for example, a simple TV that sells for roughly EUR 240 would be roughly EUR 3.45 to EUR 5.90 more expensive, or 1.5 to 2.5% of total purchase price, if PC/ABS (PC/ABS stands for Polycarbonate/Acrylonitrile Butadiene Styrene) and an alternative flame retardant were used rather than decaBDE HIPS (HIPS stands for polystyrene). However, these costs cover only the cost of raw materials for the production process and do not include the costs for any process or technology changes that may be necessary in a given case.

The costs of substitutes vary according to the type of flame retardant. The types of flame retardants most widely used today – decaBDE and antimony oxide – cost approximately 2.5 EUR/kg. In comparison, phosphorus-based alternatives, like pyromescent, cost approximately 5.9 EUR/kg and mixed phosphorus/halogenated (without penta-, octa- and decaBDEs), such as Pyrozoyl 6P, cost 5.5 EUR/kg. However due to cases where direct substitution is not possible - for example, textiles used in cars and airplanes have considerably stricter fire safety regulations - it is very complicated to assess the final costs of substitution of PBDEs. For replacement of backcoated fabric with a barrier layer or an inherently flame-retardant fiber, the solution can include costs of new materials, fabrics, foams, barrier layers, etc. In general, manufacturers tend not to divulge this information due to internal competitiveness.

The uncertainty of the estimate is quite high because technological processes in different enterprises might differ, making a switch to one or another alternative cheaper or more expensive. Also raw material prices are subject to overall market fluctuations and geographical location of the producer.

3.2.5 Secondary environmental effects

Secondary environmental effects in most cases can include increased emissions of other substances, especially the PBDE substitutes themselves. In most cases these are non-halogenated substances or phosphorus containing substances, which do not possess similar unwanted environmental effects as PBDEs. However, the impacts which might occur due to interaction of different chemical compounds should in any case be carefully evaluated before substitution.

3.2.6 Technical feasibility

As can be seen, several product groups can serve as substitutes for PBDEs, however, in some cases there are some conditions to be evaluated before implementing substitution in practise. For example, resorcinol bis-diphenylphosphate (RDP) cannot be used in pure HIPS plastics – if pure HIPS is used, also the polymer base material has to be changed for the new flame retardant to be effective. However, technical studies have shown that in most cases this is not very problematic and can be done without large capital investments. Another case is substitution with substances that are currently not under any restrictions but might be included in such lists in the future (e.g. Bisphenol A), which might result in the need for repeated substitution and unnecessary expenses.

3.2.7 Secondary socio-economic effects (including indirect costs)

Secondary socio-economic effects shouldn’t be very visible with this measure, as it has little impact on such factors as indirect costs, etc. The ready product for consumer use might get a little bit more expensive due to slightly more costly flame retardants.

3.2.8 Geographical and time scale of effects

Effects from this measure probably would not show up immediately, as it takes time for new products to enter the market and change places with existing ones. The same goes for historical pollution from waste ending up in landfills. As a result the time lag between implementing the measure and actually measureable changes in pollution concentrations can be up to a few years.

3.2.9 Political enforceability

Political enforceability is difficult to assess for this particular measure, as it is implemented by private enterprises. The driving force for implementation, however, can be a political decision to phase out particular substances, but this is usually done at international level after lengthy discussions with all involved stakeholders.

3.2.10 Cost-effectiveness analysis

Cost-effectiveness is highly dependent on different industry-specific factors, which can be numerous (type of specific production, size of installation, chemicals currently used, etc.), so it is difficult to give concrete numbers for comparison, but in general, even though at the beginning some investments might be done, substitution will, in the long run, be more cost-effective than, e.g., operating a specific waste treatment technique or paying increased fees to the municipal WWTP.

3.3 Measure 3: Urban run-off management

3.3.1 Description of source

An important source of PBDEs are urbanized areas where specific installations such as used car shredding facilities are being operated, as vehicles contain a lot of electronic equipment, plastic parts and textiles (seat covers, etc.) where PBDEs can be used as flame retardants. Parts of cars are simply shredded into smaller pieces for further recycling processes, emitting small particles of material to the surrounding area. These particles, in some cases containing PBDEs, can be flushed away by storm waters and then deposited in the environment, creating substantial source of PBDE emissions.

3.3.2 Description of measure

The solution is to implement proper run-off management from such areas. This includes a collection system for storm waters and a appropriate treatment (membrane filtration, oxidation) or connect the system to a waste water treatment facility which can assure such treatment.

3.3.3 Effectiveness

The effectiveness of any particular technique or combination of several techniques can vary depending on the volume of run-off to be treated and the concentration of PBDEs in it. In general, pre-treatment at local facilities will always give better results with reduction of pollution than chanelling waters to municipal WWTPs. Unfortunately there is a lack of data concerning treatment effectiveness of different methods for PBDE reduction.

3.3.4 Costs

The costs for such a measure can vary greatly, depending on the size and capacity of the particular installation, amount of water to be treated and/or availability of connection to municipal or industrial waste water treatment plants. As already mentioned, not many studies for run-off management have been done with regards to PBDEs removal. However, other organic pollutants have been analysed for several advanced waste water treatment techniques. The results can vary quite greatly depending on the size of the installation – as found out in the SOCOPSE project, the costs for treatment of nonlyphenols in municipal facilities can range from 0.2 to 32.8 EUR/p.e. (population equivalent) for activated carbon treatment, from 0.1 to 5.4 EUR/p.e. for membrane filtration and from 0.2 to 14.8 EUR/p.e. for oxidative treatment. For smaller scale installations which could be applied at single sites, like car-shredding facilities, these numbers could change, but the measure would still be quite efficient if loads of pollutants are high and if there are problems with more than one type of pollutant.

3.3.5 Secondary environmental effects

Secondary environmental effects from this measure can definitely be evaluated as positive, since proper water run-off management can solve problems not only with PBDEs emissions but also those of other pollutants as well, such as heavy metals and oil spills.

3.3.6 Technical feasibility

The technical implementation of this measure should not be very difficult, as a lot of companies and industrial parks which operate in large open spaces already employ collection and pre-treatment systems for their run-off waters. Also specific techniques for treatment of organic pollutants, such as PBDEs, are available in the market ready for operation.

3.3.7 Secondary socio-economic effects (including indirect costs)

Indirect costs could include increased energy consumption in case energy-intensive technical measures are employed.

3.3.8 Geographical and time scale of effects

Effects from the implementation of this measure should be visible rather soon, as loads of specific pollution discharged to municipal WWTPs and water bodies should decrease quite fast. Of course, the impact on overall pollution levels depends on how much of the total load of PBDEs is generated by such installations and how many of them actually implement the measure.

3.3.9 Political enforceability

From the policy point of view this measure should be well in line with other industry-specific goals, such as reducing environmental impacts from industrial enterprises, employing good practices in waste water management and others. As this measure may have to be integrated into an existing environmental permitting system, the question arises whether such requirements would be included in permit conditions. This is mostly a regional aspect which can vary from place to place. Especially big differences could be observed where the IPPC system is not in force, such as in the Russian Federation, where industrial lobbying is quite strong and permitting conditions could be judged by aspects other than the environmental ones.

3.3.10 Cost-effectiveness analysis

This measure most probably will require financial investments to put it into operation and possibly also additional expenses during operation, especially in cases where existing pre-treatment facilities are very simple (e.g. mechanical) or absent. The measure could become cost-effective though, if rather strict environmental fees/financial sanctions are applied for exceeding emission limit values set by municipal WWTPs or environmental inspectorates, however, as this is a national issue, no reliable comparison can be presented here. Also, as already stated, cost-efficiency rapidly increases if so called “cross-substance” effects can be achieved – meaning the savings made when several problematic pollutant groups can be satisfactorily treated with one and the same technique. Although the scientific background for this is not sufficient, their substance properties and studies for other types of organic pollutants suggest that this could very well be applicable also to PBDEs.

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IV. Perfluorinated compounds (PFCs): perfluorooctanesulfonate (PFOS) and perfluorooctanoic acid (PFOA)

4.1 Measure 1a: Substitution of PFOS in metal (chromium) plating

4.1.1 Description of source

PFOS is used as mist suppressant for non-decorative hard chromium (VI) plating and as wetting agent for use in controlled electroplating systems. These applications are exempted from the EU ban, if "the amount of PFOS released into the environment is minimised, by fully applying relevant best available techniques".

Metal plating facilities are regulated by the integrated pollution prevention and control Directive (IPPC), if the bath volume exceeds 30 m³. But there are many small and medium-sized enterprises (SMEs) in this sector, which may not be covered by the IPPCDirective.

The main applications for the final product are heavy duty engines (marine, etc.), rolling mill bearings (steel and non-ferrous metal), rollers (in paper mills), aerospace undercarriage and control components, medical equipment, automotive shock absorbers (ACEA, 2004, cited in STM BREF 2006), but also e.g. sanitary applications, such as water taps (see e.g. Fath 2008) .

PFOS is used as process aid in metal plating, but the final product does not contain any PFOS, and therefore does not contribute to urban stock via products. But many facilities seem to be indirect dischargers to MWWTPs (after pre-treatment to remove metals, see e.g. Fath 2008).

4.1.2 Description of measures

In this chapter, different options for substitution of PFOS in metal plating are reviewed: substitution with polyfluorinated substances and substitution with non fluorine substances. Both are drop in substitutes and require no change in process design

In chrome plating PFOS works by lowering the surface tension and forming a single foamy film barrier of a thickness of about 6 nm on the surface of the chromic acid bath, and thus reduces airborne loss of chromium-VI from the bath and decreases exposure of workers to this carcinogenic agent (POPRC 2010). Therefore the performance of the substitute is a crucial issue for workplace safety.

Other bottlenecks to substitution are resistance against a corrosive environment (caused by the chromium acidic electrolyte) and stability of the mist suppressing property (Bruinen de Bruin et al. (2010)). Considering the extreme chemical properties needed for this application, another bottleneck to substitutes is a significant better environmental performance than PFOS.

Adaptation of processes: According to industry information, switching to trivalent chrome electroplating (process oriented measure) is not possible for hard chrome plating (POPRC 2010). Physical barriers for aerosol forming (balls or nets), as well as adapted ventilation are being investigated, but are possibly not as effective and may therefore compromise workplace safety (POPRC 2010). Future physical solutions, in the form of adapted ventilation and other mechanical measures, would have the advantage of avoiding use and emissions of chemicals (PFOS or substitute), but further research is needed on this emerging measure. The crucial point for evaluation of this measure is workplace safety (performance against aerosol formation).

Drop-in substitutes are available for hard chrome electroplating. Often polyfluorinated substances are used as substitutes. In a recent study from Denmark (Poulsen et al. 2011) a polyfluorinated substitute was found, based on 1H,1H,2H,2H-perfluorooctane sulfonic acid (H4PFOS), equal to PFOS in performance and price. These results are also supported by a German study (Fath et al. 2010).

4.1.3 Secondary environmental effects

Polyfluorinated substances such as 1H, 1H, 2H, 2H-perfluorooctane sulfonic acid are an improvement compared to PFOS in terms of toxicity and bioaccumulation potential. This is most probably due to the lower number of highly stable C-F bonds in these molecules (van Putte 2010). The perfluorinated part of the substance is not degradable, but the non-fluorinated part can be degraded. It can be expected that the stable non degradable end product of 1H,1H,2H,2H-perfluorooctane sulfonic acid is a C6-perfluorinated substance, which is likely 10-1000 times less toxic and bioaccumulative than PFOS with its C8-chain (Poulsen et al. 2011).

In spite of this improvement compared to PFOS, the viability of polyfluorinated substances as environmentally compatible substitutes is often questioned, on account of their great stability and possible contamination of groundwater and drinking water (UBA 2009). The environmental and health performance of substitutes has to be assessed carefully in targeted studies by industry and the research community. A full assessment is outside the scope of this study. Also potentially important aspects such as increased amounts of substitute necessary for comparable performance or decreased effectiveness of end-of-pipe measures for the substitute have to be taken into account. This is especially important in cases of high emission factors.

Concerning future physical solutions the crucial point for evaluation is workplace safety (performance against aerosol formation).

4.1.4 Technical feasibility

Polyfluorinated substitutes are commercially available (reference year 2010) and reports from long term tests are available (Poulsen et al. 2011, Fath et al. 2010). Future physical solutions are an emerging measure, but targeted research is needed.

4.1.5 Secondary socio-economic effects (including indirect costs)

No secondary socio-economic effects are expected, since available substitutes seem to be moderately priced (same as PFOS based products) and can be operated as drop in. Workplace safety and health of workers seem to be maintained at high standards. For future physical solutions workplace safety (performance against aerosol formation) is crucial.

4.1.6 Geographical and time scale of effects

According to the document “Implementation of the restriction on PFOS under the Directive 2006/122/EC – electroplating applications and fire fighting foams containing PFOS stocks” issued by the European Commission on 29 January 2010, only Germany, Sweden and Finland reported use of PFOS in metal plating. The use in Denmark is confirmed by Poulsen et al. 2011. The remaining EU countries in the Baltic Sea catchment area reported that they do not use PFOS.

4.1.7 Political enforceability

National or regional regulations can be effective for individual installations and the relevant EU directive foresees a review process for the exemptions.

4.1.8 Cost-effectiveness analysis

For cost effectiveness analysis, two different scenarios for cost and effectiveness of the measure are applied, based on the approach presented in the preceding chapters. These are combined with approximated load reductions to give ranges of cost effectiveness.

The following assumptions are made: The used amount in EU is assumed to be 4 t/a (2010) , emission factor to water is assumed to be medium to high (5% - 50%) and the total load from this source in EU is estimated to be 200-2000 kg per year. Generally the uncertainties for these assumptions are high.

In scenario M1 for polyfluorinated drop in substitute the formulations containing H4PFOS have costs comparable to those with PFOS (Poulsen et al. 2011, Fath et al. 2010). The cost effectiveness of this substitution is estimated to be 100-1000 €/kg to account for potential enhanced need for surveillance.

In scenario M2 concerning future physical solutions, an estimate from Canada gives a cost of 3.9 M USD/a (5.4 M€/a) needed for improved ventilation and other mechanical measures (discounted over 25 years, Canada 2007 cited in POPRC 2007). Scaling this estimate to the EU using population as proxy results in a cost estimate of 80 M€/a. Based on this cost approximation, a cost effectiveness of 40-400 T€/kg is estimated.

4.2 Measure 1b: Substitution of PFOS/PFOA in manufacture of semi-conductors

4.2.1 Description of source

According to industry information reported in POPRC (2007) and Van Putte et al. (2010), PFOS or PFOA are used in anti-reflective coatings in combination with photoresists. In this high tech sector the performance of the chemical is crucial for product quality. Chemical formulation of photolithography products occurs under highly automated, largely closed system conditions. The same process for electronics fabrication is similarly automated, with a low volume of PFOS or PFOA used, and use of protective equipment. Chemical isolation is also an intrinsic part of quality control procedures. Emission factors are relatively low, as enclosed processes are employed and most waste is incinerated. There is no residual PFOS or PFOA compound present in manufactured products and therefore no contribution to urban stock.

4.2.2 Description of measure

According to industry information, there is no substitute available for PFOS or PFOA for manufacture of semi conductors (POPRC 2007). The costs for development of a future substitute are estimated in (POPRC 2007). It is likely that this substitute would be based on polyfluorinated substances.

4.2.3 Secondary environmental effects

As the substitute would likely be based on polyfluorinated substances, see chapter 4.1.3.

4.2.4 Technical feasibility

No substitutes are commercially available (reference year 2010). Therefore, it is an emerging measure.

4.2.5 Secondary socio-economic effects (including indirect costs)

As semi-conductors are a “high quality – high price” product, negative secondary socio-economic effects may be minor. On the other hand, global competition exists and production may be outsourced to countries with less regulation.

4.2.6 Geographical and time scale of effects

According to industry, innovation cycles may take up to 15 years.

4.2.7 Political enforceability

National or regional regulations can be effective for individual installations and the relevant EU directive foresees a review process for the exemptions.

4.2.8 Cost-effectiveness analysis

Based on information from industries (POPRC, 2007) development costs for a new photoresist system can be estimated to add up to 700 MUSD (973 M€) over a 5-year development period. The time span for analysis is 25 years and for emission reduction over 20 years. No additional running costs were assumed.

The following assumptions are made: The used amount in EU is assumed to be 0.5 – 1 t per year for PFOS in critical applications (uncritical uses ended in 2007 according to industry information) and emission to water is assumed to be approximately 5-10% (6 kg/a to 50 kg/a). As there is no information available on used amount of PFOA (other than confirmation of use (van Putte et al. 2010)), the same rough estimate as for PFOS is used for PFOA.

In scenario S1, assuming that PFOS and PFOA are substituted individually, the emission reduction (over 20 years) equals 120-1000 kg of PFOS and 120-1000 kg of PFOA. Cost effectiveness lies in the range of 1-8 M€/kg PFOS and 1-8 M€ of PFOA.

In Scenario S2, assuming that PFOS and PFOA are replaced by a single substitute, emission reduction (over 20 years) equals 240-2000 kg of PFOS and PFOA. Cost effectiveness lies in the range of 0.5-4 M€/kg of PFOS and PFOA.

4.3 Measure 1c: Substitution of PFOS/PFOA in manufacture of photographic material

4.3.1 Description of source

In the photo industry PFOS-related substances (tetraethylammonium perfluorooctane sulfonate and perfluorooctyl sulfonamidopropyl quaternary ammonium iodide) have been used in manufacturing film, paper and plates. These PFOS-related compounds function as dirt rejecters and friction control agents and they reduce surface tension and static electricity. Imaging materials that are very sensitive to light (e.g. high-speed films) benefit particularly from the properties provided by PFOS-based materials. The concentration of PFOS-related substances in coatings of films, paper and plates is in the range of 0.1–0.8 mg/cm2 (POPRC 2010). World consumption of PFOS for colour film production fell from 23 tonnes in 2000 to 8 tonnes in 2004, due to the spread of digital photography. Current annual consumption in the European Union's photographic industry is 1 tonne (POPRC 2010, RPA 2004).

According to van Putte et al. (2010), PFOS and PFOA have comparable critical applications in the photographic industry. According to I&P Europe (2010), the photo imaging industry has already discontinued all non-critical uses of PFOA-related substances. The remaining uses accounted for 2 t per year in 2008. Both PFOS and PFOA remain in the (coated) product and may therefore contribute to emissions from urban stock (e.g. via MWWTP) or from recycling facilities.

4.3.2 Description of measure

According to industry information, there is no substitute available for PFOS or PFOA in the manufacture of photographic material (POPRC 2007). The costs for future development of a substitute are estimated in POPRC (2007). It is likely that this substitute would be based on polyfluorinated substances.

Properties that alternatives must have in order to match the quality of PFOS or PFOA compounds include dynamic surface tension capability, antistatic property, solubility, photo-inactivity and stability against heat and chemicals. According to the I&P Europe - Imaging and Printing Association (Michiels 2010) PFOS/PFOA-related substances also provide important safety features due to their antistatic properties preventing product damage and controlling fire and explosion hazards. Other important properties include lack of photoactivity (no interference with the imaging process), control of surface wetting properties, and prevention of build-up of particles that can clog magnetic strip readers.

According to industry information, special products in particular, such as high-speed films and X-ray film for photo imaging for medical and industrial uses (e.g. non-destructive testing), benefit from the properties provided by PFOS- and PFOA-based materials.

4.3.3 Secondary environmental effects

As the substitute would likely be based on polyfluorinated substances, see chapter 4.1.3.

4.3.4 Technical feasibility

According to industry information compiled by van Putte at al. (2010) for the European Commission, there are as yet no substitutes ready for market for the remaining critical application and no information about the cost of substitution is available. Therefore, this substitution is considered to be an emerging measure.

4.3.5 Secondary socio-economic effects (including indirect costs)

As photographic material is a “high quality – high price” product, negative secondary socio-economic effects may be minor. On the other hand, global competition exists and production may be outsourced to countries with less regulation.

4.3.6 Geographical and time scale of effects

After the development period, the measure becomes effective in eliminating emissions of PFOS and PFOA.

4.3.7 Political enforceability

National or regional regulations can be effective for individual installations. In addition to that, the relevant EU directive foresees a review process for existing exemptions.

4.3.8 Cost-effectiveness analysis

For cost-effectiveness calculations the following assumptions were made: Use in EU is assumed to be 1 t per year for PFOS (worldwide use was 23 t in 2000 and 8 t in 2004) and 2 t per year for PFOA (POPRC 2010). Emissions to water are assumed to be 1-2 kg per year (0.1% for PFOS and PFOA, respectively) due to low factors for emission to water .

Scenario P1, which is based on previous cost estimates by RPA (2004), shows costs of US$20-40 M for reduction of roughly 15 t (worldwide in uncritical applications 2000-2004). Based on this estimate, POPRC (2007) estimates further reductions to cost more than twice as much, up to US$5 M per tonne of substitute used. For substitution of 1 t of PFOS used in the EU, the annual costs are 7 M€; for 2 t of PFOA used in EU, the annual costs are 14 M€. Cost effectiveness lies in the range of 7 M€/kg PFOS resp. PFOA.

4.4 Measure 2: Improvement of BAT and revision of BREF document for metal surface treatment

4.4.1 Description of source

See chapter 4.1.1

4.4.2 Description of measure

“Best Available Techniques” (BAT) are defined in “BAT reference documents” (BREFs) compiled by the European IPPC Bureau. BREFs are the main documents which competent authorities in member states use as a basis for issuing operating permits for installations .

The BREF document which defines current BAT in metal surface treatment dates from 2006 . As the document was written before the EU ban (Directive 2006/122/EC), it contains only few references to PFOS, giving rather general advice on PFOS emission reduction, such as closing water cycles, minimizing drag out, economic use of PFOS by measuring surface tension in the baths. In particular, the BREF document does not contain any definite reference on which waste water treatment technologies are effective for PFOS. For example AC treatment is mentioned as an option, but so is sand filtration, which is not effective for PFOS. Therefore, even with full implementation of BAT metal plating facilities can have very high emission factors for PFOS . Additionally, metal plating facilities with bath volumes <30 m³ are not IPPC regulated.

The revised BREF should include references to substitutions as well as to end-of-pipe technologies for metal plating facilities which are effective in eliminating PFOS. The issue of polyfluorinated substitutes should also be addressed, including possible EoP technologies for elimination of polyfluorinated substitutes.

4.4.3 Cost-effectiveness analysis

The cost for revision of the BREF should be covered from the funds made available for the regular review cycle; there is no information on costs for an earlier review.

4.5 Measure 3: Advanced waste water treatment - AC treatment of industrial waste water

4.5.1 Description of source

See chapter 4.1.1

4.5.2 Description of measure

Activated carbon (AC) filtration of industrial waste water from metal plating facilities is a promising EoP measure. A recent study from a large German facility (Fath 2010) recommends an AC filter for PFOS containing waste water (after chrome reduction). As part of this measure the spent AC is incinerated in order to ultimately destroy included PFOS.

4.5.3 Secondary environmental effects

Negative environmental side effects are low (additional energy use for operation of filter, transport and incineration of spent material). Elimination of other (organic) pollutants from waste water flow streams may contribute to positive environmental side effects.

4.5.4 Technical feasibility

Activated carbon (AC) filtration is a proven technology. Feasibility (and costs) depends on individual process design, such as availability of space for filtration step, existing storage tanks and separation or mixing of flow streams.

4.5.5 Secondary socio-economic effects (including indirect costs)

No information available on secondary social effects.

4.5.6 Geographical and time scale of effects

After implementation of activated carbon (AC) filtration, the measure becomes effective immediately. For geographical aspects, see chapter 4.1.6.

4.5.7 Political enforceability

Activated carbon (AC) filtration could be included in BREF document as BAT.

4.5.8 Cost-effectiveness Analysis

A recent study from a large German facility (Fath 2010) estimated investment costs of 30-60 T€ and running costs of 20-30 T€. The effectiveness of this measure is reported to be 85% (from 5.5 to 0.7 kg/a). The resulting cost effectiveness lies in the range of 5-10 T€/kg eliminated PFOS. The study stresses that costs and effectiveness are very site specific, but nevertheless the reported value is included in calculations for orientation.

4.6 Measure 4: Advanced waste water treatment - AC treatment of municipal waste water

4.6.1 Description of source

Most of the emissions from urban stock are channelled through MWWTPs, urban runoff and landfills. The types and loads of pollutants in waste water are dependent on local conditions in the urban area served, e.g. pattern of indirect dischargers, product use pattern, user behaviour and pollutant load from urban surfaces (roofs, streets etc.) in case of combined sewer systems.

The types and loads of pollutants in waste water vary greatly between different cities/districts/MWWTPs and can also vary markedly in time. Therefore, predicting type and load of pollutants at MWWTPs has a very high uncertainty.

In case of PFOS and PFOA, the uncertainty is heightened by the fact that precursor substances can be transformed to PFOS and PFOA during conventional waste water treatment. Effluent concentrations in MWWTPs therefore often exceed influent concentrations (Schultz et al. 2006, Sinclair & Kannan 2006, Becker et al. 2008, Bossi et al. 2008, Huset et al. 2008). The identity of precursors and their emission sources and environmental fate processes are largely unknown. Another factor adding to the uncertainty for this source is the rate of partitioning between water and solid phase (e.g. between effluent and sewage sludge) (Buser and Morf 2009), as well as emissions to atmosphere (e.g. during aerobic treatment) (Ahrens et al. 2011).

4.6.2 Description of measure

Activated Carbon (AC) treatment for removal of pollutants from wastewater is a proven technology. AC has a large surface area and is an effective sorbant for many substances. Different technical systems are commercially available (e.g. powder (PAC) and granular activated carbon (GAC)). The technical prerequisite for the use of AC treatment is a well-functioning MWWT with low concentrations of suspended solids and dissolved organics (BOD and COD). After waste water treatment, PFCs cannot be removed from activated carbon (LANUV 2008). Therefore, in order to avoid possible releases and to ultimately destroy PFCs, the spent activated carbon must be incinerated.

4.6.3 Effectiveness

The effectiveness of AC filters at MWWTPs for elimination of PFOS/PFOA depends on the concentration range of the pollutant, technical parameters and the matrix. At well maintained MWWTPs reduction rates of 20%-75% for PFOS and PFOA can be observed. At higher PFC concentrations, in the µg-range, reduction rates of up to >95% were observed.

4.6.4 Costs

Economic analysis in the Swiss project “StrategyMicroPoll” found costs of 10-60 € per person and year, including discounted investment costs and running costs. However, specific costs are strongly dependent on the size of the MWWTP (e.g. due to economies of scale, large WWT show low specific costs).

4.6.5 Secondary environmental effects

AC filtration at MWWTPs, which is sometimes called the 4th stage of waste water treatment, affects PFOS and PFOA emissions as well as the emissions of the other 11 HS of special concern to the Baltic Sea, which are typically present in municipal waste water in very low concentrations. AC filtration also potentially has major positive secondary environmental effects on other pollutants, such as heavy metals, organic micropollutants (which are not on the HELCOM list), pharmaceuticals and their metabolites or endocrine disrupters. But also negative secondary environmental effects can be assumed in terms of increased energy use and GHG emissions through both, construction and operation of AC treatment, and through the manufacture of activated carbon.

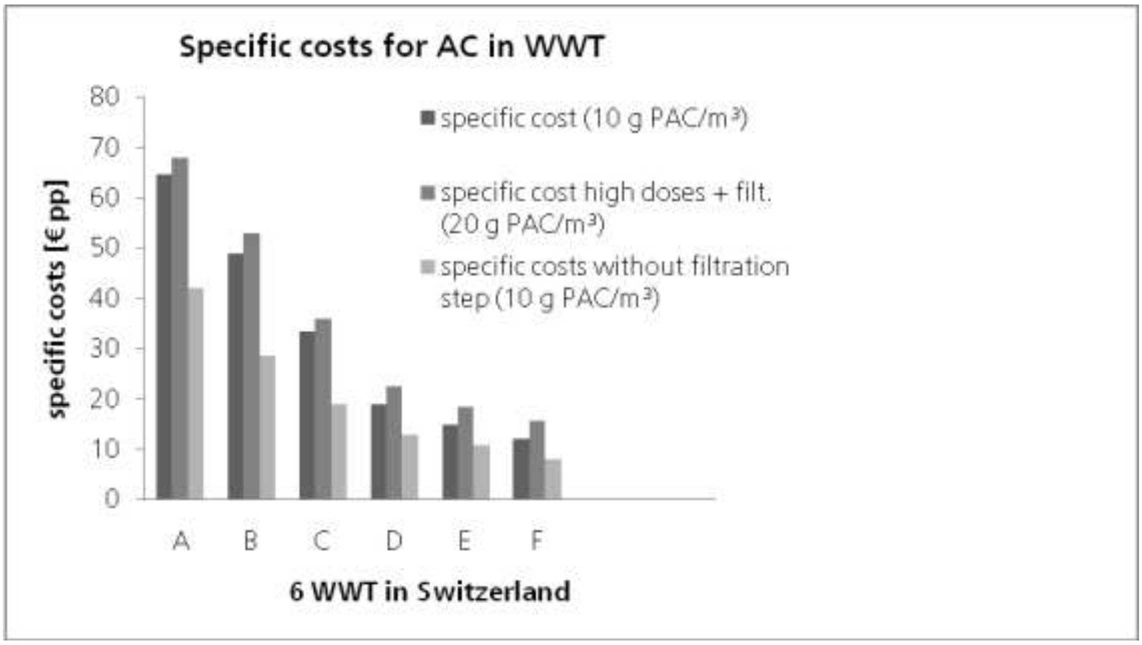

Figure 2: Specific costs per person in 6 MWWTP in Switzerland, economic data from StrategyMicroPoll

4.6.6 Technical feasibility

AC treatment is a proven technology, but the technical prerequisite is a well-functioning MWWT with low concentrations of suspended solids and dissolved organics (BOD and COD). Also skilled personnel is required for operation and maintenance (O&M) although this is likely not different from O&M of large MWWTPs.

4.6.7 Secondary socio-economic effects (including indirect costs)

If large MWWTPs are equipped with a “4th treatment stage” the question is who pays for it. One option is that the respective MWWTPs charge the cost to their clients (large MWWTPs often have lower per capita costs than smaller MWWTPs). The other option is to have the costs paid by all citizens (e.g. via taxes), as the whole community benefits from a non-toxic environment. The latter option was put into practice in Switzerland, where total costs of waste water treatment rose by 6%.

4.6.8 Geographical and time scale of effects

This measure becomes effective immediately and has a technical life span of 80 years. It seems to be suitable for large plants because of economy of scale effects. In BSR approximately 50% of the total waste water flow runs through large MWWTPs. In the context of the COHIBA project, especially MWWTPs near the shore are interesting targets.

4.6.9 Political enforceability

The political enforceability is good, for example in Switzerland it has been mandatory for large MWWTPs to eliminate hazardous substances since 2010.

4.6.10 Cost-effectiveness analysis

Two scenarios are derived to describe per-capita load ranges for PFOS and PFOA in MWWTP effluent: For PFOS a low load of 0.66 mg/cap*a and a high load of 6.9 mg/cap*a is assumed. For PFOA a low load of 0.7 mg/cap*a and a high load of 4.9 mg/cap*a is assumed. The effectiveness of AC treatment in removing PFOS and PFOA from municipal waste water is assumed to be 25%-75% .

"Scenario WW" is based on cost approximations from the Swiss project strategy MicroPoll which estimated costs for AC treatment of 15 €/cap*a for large MWWTPs . With higher efficiency of the measure (75%), cost effectiveness lies in the range of 4-40 M€/kg for PFOS and 5-38 M€/kg for PFOA. With lower efficiency of the measure (20%), costs for reduction of 1 kg are much higher, in the range of 14-150 M€/kg for PFOS and 20-143 M€/kg for PFOA.

As the measure has cross substance effects on multiple hazardous substances, the combined cost effectiveness lies in the range of 1.7-14 M€/ kg PFOS plus PFOA resulting in a higher efficiency of the measure (75%). With lower efficiency of the measure (20%), the combined cost effectiveness lies in the range of 3-40 M€/ kg PFOS plus PFOA.

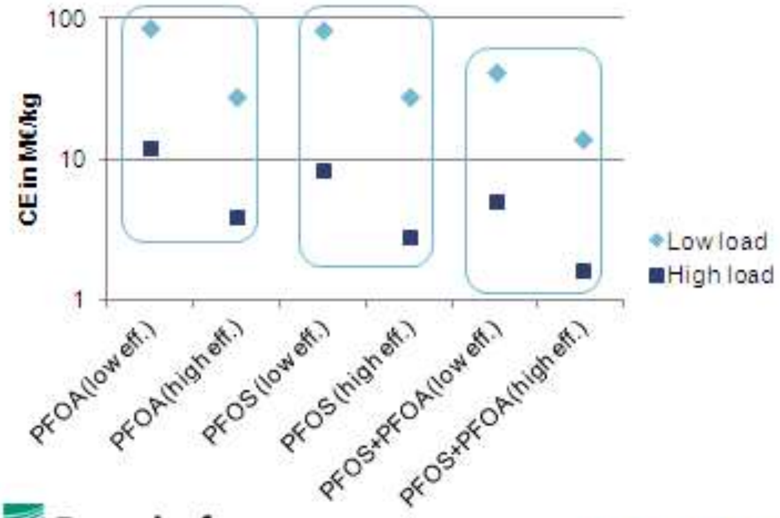

Figure 2: Comparison of cost effectiveness of measure: AC treatment of municipal waste water for elimination of PFOS, PFOA and PFOS+PFOA. Scenarios for different loads and rates of effectiveness of the measure.

4.7 Measure 5: Public awareness raising

4.7.1 Description of source

PFOA and PFOS are contained in products for "private" use in very low concentrations, for example in impregnated textiles. The total volume of these products is referred to as urban stock. Emissions from urban stock are mainly channelled through MWWTPs. The urban stock also includes products manufactured before the EU ban and voluntary industry agreements became effective. These products contain much higher concentrations of PFCs and may still contribute to emissions from "recent" urban stock, depending on technical life span of the product.

4.7.2 Description of measure

The measure "Public awareness raising" mainly targets emissions from urban stock. Consumer awareness [in the field of of hazardous substances in products is generally rather low, which can be due to the complexity of the issue. The pathways of hazardous substances in modern societies are tangled and hard to follow. Nevertheless, it can be expected that people desire to live in non-toxic cities".

Awareness raising can have several positive effects:

- Consumers prefer to buy products which are labelled as non-toxic (e.g. teflon pans without PFOA). Via reduced demand, emissions from production can be reduced.
- Consumers buy less of certain products ("chromium" water taps or car parts) as they are aware that this is a specialized product (whose production requires hazardous substances) and that the functionality of the product is not required in the foreseen application (e.g. impregnated jackets for everyday use).
- Improved public acceptance of measures which have to be paid for by the community (e.g. AC treatment of waste water, see chapter 4.5) There are no data available to quantify the costs or effectiveness of this measure.

4.8 Measure 6: Awareness raising for enterprises

4.8.1 Description of source

PFOA and PFOS are also contained in products for industrial or commercial use. Examples are mist suppressant formulations for metal plating, hydraulic fluids for aviation or fire fighting foams (until 2011). Some industrial or commercial users may discharge their waste water to public sewer systems (indirect dischargers).

4.8.2 Description of measure

Enterprises may have low awareness of how to appropriately discharge waste water or dispose of unused products as well as of possible substitutes. Therefore offering and disseminating information may contribute to reducing emissions from this source (e.g. information leaflets for enterprises or workshops).

There are no data available to quantify the costs or effectiveness of this measure.

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V. Hexabromocyclododecane (HBCDD)

5.1 Measure 1: Substitution of HBCDD as a flame retardant

5.1.1 Description of source

To render products non-flammable, flame retardants can be added to the material. HBCDD is used as a flame retardant in for example EPS and XPS insulating boards, in HIPS plastics which can be used in electronics and as a backcoating material in textiles.

5.1.2 Description of measure

To reduce the amounts of HBCDD emitted to the environment, HBCDD can be substituted with other flame retardants. Substitution can be done either by using a so called drop in substance, i.e. a substance with similar properties, or with the need to make material changes. Substitution alternatives for flame retardants and/or materials, such as EPS and XPS, HIPS and textile back-coatings, are presented below.

According to a review done by the Swedish Chemicals Agency in 2007 (KemI 2007), there were at the time of the study no commercially available alternatives to HBCDD in its use as flame retardant in EPS or XPS. A new substitute has however been developed after that and it will be phased in during 2012. For high impact polystyrene (HIPS) plastic, alternative flame retardants are available. Alternative halogenated flame retardants are decaBDE, chlorinated paraffins, brominated epoxides, decabromodiphenylethane and ethylenebis (tetrabromophtalimide) (Posner 2006). Additional information on substitutes can be found in POPRC (2011). Non-halogenated alternatives to HBCDD in HIPS, as listed by Posner (2006) and Posner et al. (2010), are resorcinol bis (biphenyl phosphate), bis phenol A bis (biphenyl phosphate), polymeric biphenyl phosphate, diphenyl cresyl phosphate, triphenyl phosphate or 1-methylethylidene) di-4,1-phenylenetetraphenyl diphosphate. For a copolymer of HIPS and polyphenylene oxide (PPO), a non-halogenated flame retardant such as triphenylphosphate can be used as a substitute. HIPS can also be substituted with another material. An alternative material to HIPS is polyethylene containing magnesium hydroxide (Posner 2006).

Intumescent systems can also be used as an alternative to HBCDD in textile back-coatings (Posner 2006). There are however doubts about durability and effectiveness. Posner (2006) also lists reactive phosphorous constituents, ammonium polyphosphate and di-ammonium phosphate as halogen-free alternatives for textile back-coating.

Another option is not to flame retard the material at all, thereby eliminating HBCDD and other flame retardants from the plastics completely (Lasses et al., 2011).

5.1.3 Effectiveness

100% removal of HBCDD is obtained when substituting HBCDD with another flame retardant.

5.1.4 Costs

There is no information on costs related to switching to HBCDD alternatives in EPS/XPS. Switching from the use of a brominated flame retardant (not decaBDE) to a phosphorous flame retardant such as triphenylphosphate would mean an increased cost of EUR 4-5 for a full (average) TV set (Danish EPA 2006 quoted in KemI 2007).

5.1.5 Secondary environmental effects

When substituting HBCDD with another flame retardant, it is important to keep in mind possible environmental effects of the substitute. Possible environmental effects caused by emissions of the suggested substitutes have not been evaluated for the purposes of this document. According to Morose (2006), further research on environmental and human health effects caused by substitutes to HBCDD is needed in order to ensure that the substitutes are less hazardous than the chemical that they are replacing.

5.1.6 *Technical feasibility*

For the above mentioned substitutes, technologies for replacing HBCDD are available. For substitution of HBCDD in EPS and XPS, a drop-in substitute will be available for testing. It is expected that demand will be fully satisfied in 2015. The substitutes need to be tested and approved by EPS producers, polystyrene foam producers and in some countries national bodies responsible for authorizing the sale of building materials.

5.2 Measure 2: Changing of product material

5.2.1 *Description of source*

Rigid EPS and XPS boards are used to insulate floors, walls and/or roofs of buildings. To allow buildings to withstand fire, these insulation boards can be flame retarded. According to the Swedish Chemicals Agency (KemI 2007), the only commercially available flame retardant for use in EPS and XPS is HBCDD. According to ECB (2008), 0.7 percent by weight is the maximum amount of HBCDD used in EPS insulating boards. Thus, an insulating board of 1 m2 that is 60 mm thick weighs approximately 0.4 kg and contains a maximum of 3 g of HBCDD. However, views on the need for flame retardants in polystyrene insulation differ between BSR countries.

5.2.2 *Description of measure*

To decrease the amount of HBCDD in buildings, flame retarded EPS and XPS can be replaced by other materials, for example materials that are flame retarded with flame retardants other than HBCDD or with materials that are inherently non-flammable and therefore do not need any flame retardants.

An alternative flame retardant material for insulation of buildings is polyurethane (PUR) rigid foam, while mineral based products such as mineral wool are an example of an inherently non-flammable material (KemI 2007). However, mineral wool is not applicable for use in floors since it lacks the structural stability of EPS, XPS and PUR. Posner et al. (2010) also mention phenolic foam, blanket insulation (may also be flame retarded) and loose-fill insulation as alternative materials.

A third option is to redesign the buildings and instead use insulation boards that are flammable but are not flame retarded in combination with building techniques that reduce the risk of the insulating material catching fire, e.g. by encapsulating the insulating material in fire resistant materials such as gypsum board, gypsum or cement plasters, perlite board, spray-applied cellulose, mineral fibre, or gypsum coatings, and select plywoods (Posner et al. 2010). However, the option of redesigning buildings has not been studied further here due to a lack of data.

Additional information about alternatives to flame retarded EPS in buildings can be found the report by Lasses et al. (2011).

5.2.3 *Effectiveness*

By substituting flame retarded EPS and XPS with other materials, a reduction of the use of insulating boards flame retarded with HBCDD can be achieved.

5.2.4 *Costs*

The German UBA calculated relative (material) costs of the different insulating materials as follows (UBA 2000 quoted in KemI 2007):

- EPS: 100%
- Mineral wool: 130% (150% - according to Polish estimates)
- PUR: 280%
- XPS: 300%

Costs for EPS insulating boards on the Swedish market in 2011 are approximately EUR 6-11/m2 insulating board. The difference in price reflects different thicknesses of the material as well as differences in material qualities. More recent information on alternative materials for use in buildings can be found in Lasses et al. (2011).

5.2.5 Secondary environmental effects

When replacing EPS and XPS with another flame retarded material, there is a risk that the flame retardant in that material may be emitted to the environment instead. Environmental effects due to emissions of substitute may occur. However, no assessments of the environmental performance of the alternative materials have been found in the reviewed literature. Health concerns are however related to the use of some alternative materials such as phenolic resin monomers produced with formaldehyde.

5.2.6 Technical feasibility

Structurally, it is possible to replace EPS and XPS with PUR. In the case of mineral wool, material substitution is not possible when the insulating board is intended for use in floors since mineral wool lacks the necessary structural stability. The technical feasibility of replacing flame retarded EPS with other materials have been studied in more detail in Lasses et al. (2011).

5.2.7 Cost-effectiveness analysis

As mentioned above, the maximum amount of HBCDD in an EPS board 1 m2 in size and 60 mm thick is approximately 3 g. Using this together with the relative costs compiled by UBA (2000), see section 5.2.4, the costs for reducing emissions of HBCDD from EPS insulating boards can be estimated.

In Table 1, the cost increase per square meter of insulating board and the cost-effectiveness for reducing emissions of HBCDD from EPS insulating boards by material substitution is presented for an insulating board with a thickness of 60 mm. In the calculations, it has been assumed that all HBCDD in an insulating board is emitted to the environment sometime during its lifetime (including both use and waste management) due to the persistency of HBCDD, unless the insulating board is incinerated. If the insulating board is incinerated, the HBCDD is instead assumed to be destroyed. This may result in an overestimation of emissions.

Table 1: Costs for substituting EPS boards as building materials

Substitutes	Cost increase (EUR/m^2)	Cost-effectiveness (EUR /kg reduced HBCDD)
Mineral wool	1.7-3.4	560-1100
PUR foam	16-32	5300-11000

If XPS is replaced with mineral wool or PUR, a cost reduction would be achieved since both mineral wool and PUR cost less than XPS.

5.3 <u>Measure 3:</u> Municipal waste water treatment

5.3.1 Description of source

HBCDD is commonly found in urban waste water due to diffuse spreading from the use of products and plastic materials in society. Lilja et al (2010) detected HBCDD in 7 out of 7 waste water samples taken in Sweden, and Fjeld et al (2007) detected HBCDD in 91 out of 246 waste water samples taken in Norway. Urban waste water treatment plants may therefore be an important source to the recipient load of HBCDD from diffuse spreading.

Average concentrations of HBCDD in influent waste water measured at five different Norwegian waste water treatment plants (Fjeld et al. 2007) range between 1.4-22.5 ng/l. HBCDD concentrations in effluent have been measured in the screening study performed within COHIBA WP3 (Nakari et al., 2011). The results show effluent concentrations ranging between 1-68 ng/l with a median of 3.8 ng/l. Effluent concentrations have also been measured by Fjeld et al. (2007) and Lilja et al (2010), with concentrations of 0.9-1.6 and 0.05-0.27 ng/l respectively.

5.3.2 Description of measure

Urban waste water treatment is covered by the Urban Waste Water Directive (Council Directive 91/271/EEC), where it is stated that waste water should be collected and treated in all agglomerations of more than 2000 p.e. (population equivalents).

Urban waste water treatment plants generally use mechanical and chemical treatment as well as biological treatment to reduce pollutants in waste water. Urban waste water treatment plants significantly reduce the concentration of HBCDD in the effluent water by partitioning the HBCDD to the sludge. Since degradation of HBCDD in water is very slow, mechanical, chemical and biological waste water treatment does not contribute substantially to the degradation of HBCDD in the water phase (Nordic Council, 2008). Studies of waste water treatment plants in Norway have not shown any large differences in removal of HBCDD from the water phase in plants with mechanical and chemical treatment compared to plants that also had biological treatment (Fjeld et al, 2007).

It is also possible to use advanced waste water treatment techniques, such as AC (activated carbon) treatment to further increase the removal of HBCDD from the effluent. However, this measure is not included here.

5.3.3 Effectiveness

In waste water treatment plants with mechanical, chemical and biological treatment, approximately 70-90% of the HBCDD is adsorbed to sludge, while the remaining HBCDD is emitted to receiving waters via waste water effluents (Fjeld et al, 2007; Nordic Council, 2008).

5.3.4 Costs

This parameter has not been assessed in the present study.

5.3.5 Secondary environmental effects

Urban waste water treatment has a number of secondary environmental effects, many of them positive. For example, besides reducing the concentration of HBCDD in the effluent, urban waste water treatment plants also reduce concentrations of phosphorus and nitrogen, which reduces eutrophication. Urban waste water treatment plants also reduce concentrations of chemicals other than HBCDD. Negative secondary environmental effects can be connected to the use of energy and chemicals during waste water treatment process.

5.3.6 Cost-effectiveness analysis

This parameter has not been assessed in the present study.

5.4 Measure 4: Industrial waste water treatment

5.4.1 Description of source

Industrial waste water can contain high concentrations of HBCDD, for example waste water from industries processing plastic materials such as EPS and XPS. Concentrations of HBCDD in effluent from industrial sites have been measured in the COHIBA project (Nakari et al., 2011). In effluent from one industry, a HBCDD concentration of 110 ng/l was measured.

5.4.2 Description of measure

The same technologies as are used for municipal waste water treatment can also be used to treat industrial waste water. Industrial waste water can contain high concentrations of HBCDD. By separating the sludge at site before sending the waste water to municipal waste water treatment plants, the load of HBCDD to the municipal WWTP can be lowered.

5.4.3 Effectiveness

In waste water treatment plants with mechanical, chemical and biological treatment, approximately 70-90% of the HBCDD is adsorbed to sludge, while the remaining HBCDD is emitted to receiving waters via waste water effluents (Fjeld et al, 2007; Nordic Council, 2008).

5.4.4 Costs

Costs for installing a small waste water treatment plant, dimensioned for 10 000 p.e., which can be used at an industrial site for example, are presented here. The data were taken from a Swedish WWTP that was constructed during 2010 (Kungsbacka Kommun 2007). The studied WWTP has mechanical and chemical treatment and separates sludge by sedimentation. Investment costs were available, running costs however were not available for the studied WWTP as it has only been in operation for a short period. Instead, running costs were taken from another Swedish WWTP and scaled up or down, depending on the number of connected "population equivalents" (PE). As a result, the cost for treating one m3 of waste water was EUR 0.33. For more detailed information about these costs, see Appendix B.

Due to the large uncertainties introduced by the way costs were estimated for this case the results should be interpreted with caution and only serve as an illustration of the possibilities of installing sludge separation at an industry.

5.4.5 Secondary environmental effects

Industrial waste water treatment has a number of secondary environmental effects, many of them positive. For example, besides reducing the concentration of HBCDD in the effluent, industrial waste water treatment plants also reduce concentrations of phosphorus and nitrogen which reduces eutrophication. Industrial waste water treatment plants also reduce concentrations of chemicals other than HBCDD. Negative secondary environmental effects can be connected to the use of energy and chemicals during the waste water treatment process.

5.4.6 Cost-effectiveness analysis

The cost effectiveness of reducing HBCDD by separating sludge from industrial waste water effluent was estimated based on two different removal efficiencies (70% and 90%) and a HBCDD concentration of 110 ng/l. The resulting cost-effectiveness is estimated to be in the range of EUR 3 300 000-4 300 000/kg reduced HBCDD. This however leaves the industry with volumes of HBCDD containing sludge. For this sludge there are various treatment options with varying efficiencies and costs (see sections 5.4).

This alternative complies with the polluter pays principle (PPP) and does not therefore contribute to increased direct costs for municipalities running M-WWTPs.

Table 2: Cost-effectiveness of on-site separation of sludge in a small mechanical and chemical WWTP at an industrial site

Removal efficiency	Cost effectiveness (M €/kg reduced HBCDD)
90%	3.3
70%	4.3

5.5 Measure 5: Sludge treatment - anaerobic digestion

5.5.1 Description of source

HBCDD is commonly found in urban waste water due to diffuse spreading from the use of products and plastic materials in society. During waste water treatment, up to 90% of HBCDD partitions to sludge. It is therefore important to treat the sludge in an appropriate way to minimize emissions to the environment.

Concentrations of HBCDD in sludge from municipal waste water treatment plants located in the Baltic Sea Region have been measured within the COHIBA project, WP3 (Nakari et al., 2011). Measured concentrations ranged between 9.5-210 ng/g dw with a median of 130 ng/g dw. As these concentrations are specific to the Baltic Sea Region, they will later be used for performing cost-effectiveness calculations.

5.5.2 Description of measure

In anaerobic sludge digestion, sludge is fed into a digester where microorganisms degrade the biological material. Anaerobic means that the entire process takes place in the absence of oxygen. The retention time of the sludge in the digester is approximately 15-30 days (Swedish EPA, 2008).

Anaerobic digestion of sludge has been shown to degrade HBCDD by microbially mediated degradation. Experiments of anaerobic degradation of HBCDD have shown half-lives of 0.66 - 5 days (Gerecke et al, 2006; Hunzicker et al. 2004 (quoted in Gerecke et al (2006)); Fjeld et al, 2007). Full scale tests have also shown anaerobic degradation of HBCDD. The expected lifetime of an anaerobic digester is approximately 50 years.

5.5.3 Effectiveness

At a Swiss WWTP, the amount of HBCDD in sludge was reduced by approximately 40% after 28 days in a mesophilic digester (Gerecke et al, 2006). However, degradation rates can vary significantly between different digestion reactors due to differences in anaerobic environment and prevailing redox conditions (Gerecke et al, 2006). The potential which anaerobic digestion has for removal of HBCDD can therefore be assumed to be somewhere between 0-40%.

5.5.4 Costs

In order to collect economic data on anaerobic sludge digestion, investment costs and running costs of two different Swedish WWTPs with anaerobic digesters of different sizes were studied. The capacities of these two digesters are approximately 14 000 tonnes dw (dry weight) per year and 2300 tons dw per year respectively. The estimated lifetime of both digesters is 50 years (detailed information regarding investment costs and running costs can be found in Appendix B). The cost for digesting one tonne of sludge was calculated based on the sum of all investment costs and running costs during the lifetime of the digesters divided by the total amount of digested sludge. For the large digester, the cost for treating one tonne of sludge was approximately EUR 67/tonne dw, while for the smaller digester the cost was approximately EUR 170/tonne dw (see Table 3).

Table 3: Expected lifetime, capacity and costs for two studied anaerobic digesters

	Large WWTP	Small WWTP
Lifetime (years)	50	50
Yearly digested sludge (tonne dw/year)	13 725	2 340
Total amount of digested sludge during lifetime (tonne dw/lifetime)	686 250	117 000
Cost (€/tonne dw)	67	170

5.5.5 Secondary environmental effects

Besides degrading HBCDD, anaerobic digestion of sludge also generates methane gas which can be used as fuel, electricity or for heating, thereby possibly replacing fossil fuels. However, there is a risk of leakage of methane during digestion, which can contribute to global warming since methane is a greenhouse gas. The amount of digested sludge that remains after anaerobic digestion is smaller than the amount of sludge used as input to the digester, which reduces the amount of waste generated by waste water treatment plants.

5.5.6 Technical feasibility

Anaerobic digestion of sludge is an existing technology which is in use at several waste water treatment plants, for example in Sweden and Norway. An anaerobic digester can be added to an existing mechanical/chemical/biological waste water treatment plant without the need for major rebuilding of the existing plant.

5.5.7 Cost-effectiveness analysis

Table 4 and Table 5 below present cost-effectiveness data for two anaerobic digesters with regards to reduction of HBCDD levels. Three different removal efficiencies (5%, 20% and 40% reduction) as well as three different concentrations of HBCDD in sludge (low: 9.5 ng/g dw, median: 130 ng/g dw and high: 210 ng/g dw) were used for cost-effectiveness calculations in order to cover the likely span of costs for reducing HBCDD.

The lowest reduction potential in combination with the lowest concentration of HBCDD gives rise to the highest costs and vice versa. The cost-effectiveness for removing HBCDD by anaerobic digestion at a small WWTP would be in the range of EUR 2-368 M/kg reduced HBCDD while for large WWTP the cost-effectiveness would be in the range of EUR 1 000 000-142 000 000 /kg reduced HBCDD depending on removal efficiency and concentration.

These figures do not include incomes from the production of biogas. Including these incomes would improve the cost-effectiveness of removing HBCDD by anaerobic digestion.

Table 4: Cost-effectiveness for an anaerobic digester in a small WWTP

Cost-effectiveness (M €/kg reduced HBCDD)				
		Concentration HBCDD		
		Low	Median	High
Removal efficiency	40%	46	3	2
	20%	92	7	4
	5%	368	27	17

Table 5: Cost-effectiveness for an anaerobic digester in a large WWTP

Cost-effectiveness (M €/kg reduced HBCDD)				
		Concentration HBCDD		
		Low	Median	High
Removal efficiency	40%	18	1	1
	20%	36	3	2
	5%	142	10	6

5.6 Measure 6: Sludge treatment - thermal hydrolysis and anaerobic digestion

5.6.1 Description of source

For description of the source please see Section 5.5.1

5.6.2 Description of measure

Thermal hydrolysis can be used as a pre-treatment step before anaerobic digestion of sewage sludge. During thermal hydrolysis, the sludge is first heated to approximately 165˚C for 20-30 minutes, followed by rapid transportation into a flash tank which results in cell destruction caused by pressure drop (Cambi, 2011). This yields an easily digestible feed for micro-organisms in the anaerobic digestion process. The lifetime of the thermal hydrolysis plant and the anaerobic digester is estimated to be 50 years.

5.6.3 Effectiveness

In a Norwegian study by Fjeld et al (2007), a WWTP using thermal hydrolysis in combination with anaerobic digestion of sludge showed a high degradation of HBCDD: approximately 90% of the HBCDD in the sludge was degraded using this technique. No other studies of HBCDD degradation rates during thermal hydrolysis and anaerobic digestion have been found.

5.6.4 Costs

A Norwegian study by Fjærgård & Sander (2000) was used to obtain economic data on thermal hydrolysis and anaerobic digestion. In this study, investment costs and running costs for a thermal hydrolysis plant and an anaerobic digester at a Norwegian WWTP serving approximately 65 000 p.e. were determined. According to Fjærgård & Sander (2000), the cost for treating one tonne of sludge was approximately EUR 350/tonne dw (see Appendix B for more detailed information).

5.6.5 Secondary environmental effects

Like anaerobic digestion, thermal hydrolysis followed by anaerobic digestion generates methane gas and reduces the amount of sludge generated by the waste water treatment plant. With thermal hydrolysis, the cell structure of the feed disintegrates before reaching the digester, which generates a more biodegradable feed. This yields more biogas and less digestate compared to treatment by anaerobic digestion only.

5.6.6 Technical feasibility

Thermal hydrolysis in combination with anaerobic digestion is an emerging technology which has only been installed at a limited number of waste water treatment plants. However, like anaerobic digestion, this technology can be added to an existing waste water treatment plant.

5.6.7 Cost-effectiveness analysis

As was done for anaerobic digestion, cost-effectiveness for thermal hydrolysis and anaerobic digestion were calculated for three different removal efficiencies (5%, 45% and 90%) as well as for three different concentrations of HBCDD (low: 9.5 ng/g dw, median: 130 ng/g dw and high: 210 ng/g dw).

The lowest reduction potential in combination with the lowest concentration of HBCDD gives rise to the highest costs and vice versa. The cost-effectiveness for removing HBCDD by using thermal hydrolysis in combination with anaerobic digestion is in the range of EUR 4 000 000-730 000 000 /kg reduced HBCDD (see Table 6).

This process also produces biogas as a byproduct, which can generate an income. Compared to anaerobic digestion, more biogas is generally produced per tonne of digested sludge when using a combination of thermal hydrolysis and anaerobic digestion. The cost efficiency of reducing HBCDD with thermal hydrolysis could therefore be improved if incomes from biogas production were included.

Table 6: Cost-effectiveness of thermal hydrolysis and anaerobic digestion at a medium size WWTP

Cost-effectiveness (M €/kg reduced HBCDD)				
		Concentration HBCDD		
		Low	Median	High
Removal efficiency	90%	41	3	2
	45%	81	6	4
	5%	729	53	33

5.7 Measure 7: Waste management – Controlled incineration

5.7.1 Description of source

This measure is commonly used for the treatment of municipal solid waste. The load of HBCDD in municipal waste and the absolute amount of HBCDD that is decomposed as a result of incineration of one tonne of waste is difficult to determine. The amount of HBCDD in the incinerated waste depends largely on which waste fractions are burned. If only waste from building material is burned, the load of HBCDD is higher than if a mixture of household and building waste is burned.

5.7.2 Description of measure

During incineration, the waste is heated to approximately 850-950˚C (ECB European Chemicals Bureau, 2008). Since HBCDD decomposes at 190˚C; the majority of the HBCDD in the waste is decomposed during incineration (Posner et al, 2010). There is a risk of brominated dioxins and furans formation. The flue gases are treated to reduce the amounts of hazardous substances before being emitted.

5.7.3 Effectiveness

Emissions of HBCDD from incineration plants are believed to be negligible. However, no exact figures on removal potential have been found in the literature. It is assumed here that 90-100% of the HBCDD is decomposed during incineration.

5.7.4 Costs

This parameter has not been assessed in the present study.

5.7.5 Secondary environmental effects

During incineration of bromine containing waste, formation of dioxins can occur. Only a few studies on controlled incineration of materials containing HBCDD have been performed (Brenner, 2003, quoted in Posner et al 2010). These studies showed that very low amounts of dioxins and dibenzofurans are formed during the incineration of HBCDD containing material. In modern incineration facilities, emissions of dioxins can be reduced using post combustion steps and flue gas treatment systems (Funcke & Hemminghaus, 1997, quoted in Lundstedt, 2011).

5.7.6 Cost-effectiveness analysis

This parameter has not been assessed in the present study.

5.8 Measure 8: Waste management – Controlled landfilling

5.8.1 Description of source

Controlled landfilling is another waste management alternative. When material containing HBCDD, such as polystyrene, is landfilled, the HBCDD may be emitted via landfill leachate, it may be vaporized or it may become degraded in the landfill.

However, studies of emissions of HBCDD to air from landfills have shown that landfills are not an important source of HBCDD emissions to the atmosphere (Andersson et al., 2012).

Concentrations of HBCDD in landfill leachate have been measured at landfills in the Baltic Sea Region in the COHIBA project, WP3 (Nakari et al., 2011). Measured concentrations ranged between <0.1-75 ng/l with a median of 3 ng/l.

5.8.2 Description of measure

To reduce the emissions of HBCDD from landfills, landfill leachate can be collected and treated in a waste water treatment plant.

5.8.3 Effectiveness

The effectiveness of this method is the same as for waste water treatment if the leachate is collected and treated in a waste water treatment plant, i.e. 70-90% removal efficiency (see section 5.3).

5.8.4 Costs

This parameter has not been assessed in the present study.

5.8.5 Secondary environmental effects

Landfill leachate, which is generated when water infiltrates the landfill, can contain hazardous substances which can cause negative environmental effects if the leachate is not collected and properly treated. Methane may also be emitted from landfills due to degradation of the material in the landfill. As methane is a greenhouse gas, it contributes to global warming if emitted to the atmosphere.

5.8.6 Technical feasibility

Controlled landfilling is a technology that is proven and available.

5.8.7 Cost-effectiveness analysis

This parameter has not been assessed in the present study.

5.9 Measure 9: Waste management – Demolition of buildings

5.9.1 Description of source

As HBCDD is used as a flame retardant in insulating boards, HBCDD can be emitted during the demolition of buildings. The EU risk assessment (ECB, 2008) has identified the demolition of buildings as a potentially important source of HBCDD to the environment. During demolition, particles and smaller fragments of the materials in the building are generated. These particles and small fragments, consisting of flame retarded EPS or XPS, may cause the release of HBCDD to the environment.

The amount of dust generated differs from material to material. Studies have shown that the breaking of XPS insulating boards generates less dust than the breaking of EPS insulating boards (ECB, 2008). The amount of dust generated is also dependent on the demolition method used.

5.9.2 Description of measure

There are a number of different building demolition techniques available, such as manual deconstruction, implosion by the use of explosives or demolition by the use of a wrecking ball or a crane. Demolition using explosives or a crane and wrecking ball generates the most particles while manual deconstruction generates the least particles. Manual deconstruction is therefore the preferred alternative in order to reduce emissions of HBCDD.

To further minimize emissions from demolition of buildings, it is also of importance to treat the generated waste appropriately, for example by storing HBCDD containing waste in covered containers to minimize spreading of particles.

During waste management, recycling of HBCDD containing materials is not preferable from an environmental perspective, since recycling often involves shredding which generates particles (Posner et al 2010). In addition, recycling EPS/XPS with HBCDD will lead to contamination of the recyclate. Incineration is likely the preferred alternative as much of the HBCDD is destroyed during incineration.

5.9.3 Costs

This parameter has not been assessed in the present study.

5.9.4 Cost-effectiveness analysis

This parameter has not been assessed in the present study.

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VI. Nonylphenol (NP) and Nonylphenol ethoxylates (NPE)

6.1 Measure 1a: Substitution of NPE in cleaning

6.1.1 Description of source

NPEs are non-ionic surfactants used in the manufacture of cleaning products applied by industry, other professionals and households. There is a vast range of products which can be classified as institutional and industrial cleaning products including:

- Janitorial products such as hard surface cleaners, cream cleansers, hand gels and cleaners, floor cleaners and strippers, bactericidal cleaners, disinfectants, manual dishwashing liquids, carpet cleaners and acid toilet cleaners;
- Automotive products such as traffic film remover, wash & wax, car shampoos, windscreen washes, pressure cleaners, chrome cleaners, rinse formulations, engine degreasers and upholstery cleaners;
- Products for the food, dairy and catering industries such as machine dishwashing products, rinse aids, bottle washing, steam cleaners, cleaning-in-place and iodophors;
- Industrial laundry detergents, both powder and liquid products;
- Products for metal cleaning and preparation including immersion cleaners, spray cleaners, solvent degreasers, electrolytic cleaners and rinsing formulations;
- Specialist cleaners such as oil spill dispersants, rig wash, quick-break emulsifiers,abattoir cleaners, general foam cleaners, micro-emulsion cleaners, concrete cleaners/graffiti removers, paint stripper/brush cleaners; and
- Domestic detergents.

The use of nonylphenol and nonylphenol ethoxylates has been restricted within the EU since 1.1.2005 due to Directive 2003/53/EC (currently REACH, Annex XVII) in the following processes:

- Industrial and institutional cleaning except controlled closed dry cleaning systems, where the washing liquid is recycled or incinerated and cleaning systems with special treatment, where the washing liquid is recycled or incinerated (i.e. conditional ban)
- Domestic cleaning (total ban)

However, manufacture and export of cleaning products containing NP and NPE is still allowed in EU. In addition to that, it seems that NPEs are still used in industrial, institutional and domestic cleaning (manufacture and/or use) in the Baltic Sea catchment area (at least in Estonia, Finland, Latvia, Poland and Sweden). Private cleaning use of NPE has been identified in Poland and it is possible also in Finland. It seems that NPE is much more commonly used as a surfactant in cleaning products than NP (COHIBA WP4 draft national SFAs on NP&NPE). Additionally, automotive products containing NPEs may also be used in Denmark (Kjølholt et al. 2007).

If all existing regulations are followed, NPE is not discharged to the environment or to WWTP. Nevertheless, COHIBA WP4 national SFAs show that NP and NPE emissions via industrial cleaning are still significant. Thus, this conditional use restriction on industrial, institutional and domestic cleaning does not seem to be efficient.

6.1.2 Description of measure

The possible substitutes of NPE in detergents and cleaning agents are mixtures of alcohol ethoxylates (RPA 1999 & Lecloux 2010). In most cases NPEs can be replaced with alternative substances, many of which have been known for some time but not chosen due to their higher costs and lower formulation versatility. Now, improved formulation techniques are allowing companies to utilise these alternatives which are almost exclusively alcohol ethoxylates, generally either:

* C9-11 linear alcohol ethoxylates;

* C13-15 linear alcohol ethoxylates; or

* C13-15 isotridecanol ethoxylates (branched).

In the industrial and institutional sector, a typical substitute is Synperonic 13/9 (trade name) in which the parent alcohol is branched tri-decanol (i.e. R is $C_{13}H_{27}$) and the value n – representing the number of ethylene oxide units – is nine.

6.1.3 Costs

The estimated costs for substitution of NPE with alcohol ethoxylates in the industrial cleaning sector at an EU level (RPA 1999) based on CESIO data (Alkylphenol Ethoxylates Task Force of CESIO, the CEFIC sector group representing the surfactants industry) suggest that a complete ban on NPEs and OPEs would cost the industrial cleaning sector a total of EUR 156 million (unit costs of EUR 5.6 / kg used NPE/OPE), where this includes onetime reformulation and commercialisation costs (total EUR 137 million; unit costs of EUR 4.9 / kg used NPE/OPE) and change in raw materials costs (total EUR 19 million; unit costs of EUR 0.68 / kg used NPE/OPE) for one year resulting from the use of alternatives based on estimated consumption of 28 000 tons NPE/OPE /year (year 1995).

6.1.4 Geographical and time scale of effects

The substitution of NPEs is assumed to take between less than one year and a maximum of seven years, depending on the application. In most cases, substitution can be achieved in around two years (RPA 1999). Owing to concerns over the confidentiality of data in this highly competitive sector, companies completing our survey were unwilling to provide data on the change in costs arising from the move to substitutes (RPA 1999). Therefore, cost data provided by CEFIC/CESIO (RPA 1999) is used in the CEA below.

6.1.5 Cost-Effectiveness Analysis

Possible substitutes of NPE in detergents and cleaning agents are reported by some as be-ing less cost-effective than NPEs in terms of delivering the desired performance from a product. Thus, in some cases, manufacturers may need to accept inferior performance coupled with additional costs. However, other companies surveyed as part of this study have indicated that the performance of products based on the substitutes will be compara-ble and that there will be no effective loss of quality (RPA 1999).

The used amount in industrial and institutional as well as in domestic cleaning in Finland is reported to be 0 tons NP/year and 21 tons NPE / year (Mehtonen 2011) and emissions to waste water through the use of cleaning products in Finland is 9100 kg NPE /year (Mehtonen 2011).

The used amount in manufacture of cleaning products in Finland are 0 tons NP/year and 1.2 tons NPE / year (Mehtonen 2011).

Emissions to waste water from manufacture of cleaning products in Finland are assumed to be minor (NPE emissions of 4 kg/year to waste water and 0.01 kg/year to air, (Mehtonen 2011) compared to emissions from use of cleaning products).

Emissions to land from use of cleaning products in Finland: 2000 kg NPE /year (Mehtonen 2011).

Emissions to surface water via (all) waste water treatment plant effluents due to use of cleaning products in Finland: 455-910 kg NP&NPE /year with assumption of 90-95% reduction efficiency in waste water treatment plant.

Finland, Cost Scenario C1:

The total one-off investment costs to substitute NPE with alcohol ethoxylates in all cleaning uses (industrial, institutional & domestic cleaning) in Finland are estimated to be EUR 183 000 with the following assumptions;

- Used amount in industrial, institutional and domestic cleaning in Finland: 21 tons NPE / year
- One-off unit costs of EUR 8.7 / kg used NPE (discounted with annual inflation rate 3% from year 1997 to 2011) is expected for the substitution for all cleaning product types
- No additional operational and maintenance costs (i.e. assumed to be same as for NPE)
- No end of life costs
- Substitution costs for domestic and institutional cleaning are same as for industrial cleaning
- Manufacture of cleaning products in Finland is not taken into account in CEA calculations, because emissions from manufacture are assumed to be minor compared to emissions from use. The used cleaning products are mainly imported from other countries □ emissions reductions are directed to Finland but costs to firms in those countries manufactured the cleaning products.
- It will take 5 years to fully substitute NPE in all cleaning product types
- Time span for the analysis is 25 years

Cost Scenario	Costs over 25 a [MEUR]	Emission reduction to surface water [kg/25 a]		Cost Effectiveness [EUR/kg]	
		min.	max.	max.	min.
C1	0.18	9100	18 200	10	20

6.2 <u>Measure 1b</u>: Substitution of NP and NPE in paints

6.2.1 Description of source

NP and NPE are used as stabilisers and emulsifying agents in the manufacture of paints and paint resins applied by industry, professionals and households. NPE is used in the preparation of paint resin (polyvinyl acetates - PVA) and also as a paint mixture stabiliser. Typical formulations contain 0.6-3% NPE. NPEs are mainly used in decorative emulsions but small volumes are also used in other applications such as water-based 'refinish' paints for vehicle re-coating. In decorative emulsions nonylphenol ethoxylates are used in the manufacture of the emulsion and directly as emulsifiers and dispersants in water-based paints. Nonylphenol ethoxylates are most widely used as dispersants in coloured emulsions. The residual NP content in the emulsion polymer is considered negligible (EU RAR 2002).

At the moment the use of NP and NPE in paints is not restricted at the EU level. Thus, NP and NPE are discharged to sewer systems of municipal WWTPs and industrial WWTPs. There are no direct releases to surface water, air and land.

6.2.2 Description of measure

The main chemicals providing the basis for the substitutes in paints are alcohol ethoxylates. Already in the end of 1990s in UK (RPA 1999) some of the paint companies had substituted NPE with alcohol ethoxylates at least in some of the end products.

A Danish study has explored the possibilities of substituting NPEs in paints. The study identified a number of possible substitutes for nonylphenol ethoxylates in paints such as styrene/maleic anhydride polymer, secondary ethoxylated alcohol (C12-C14) and ethoxylated acetylenic alcohol. All of the alkylphenol ethoxylates (APEOs) investigated could in principle be substituted by alternative substances, but in some cases it would be a time-consuming process. Thus, it is technically possible to replace nonylphenol ethoxylates in paints etc., however, it is not clear, whether these possibilities are presently sufficiently developed for all uses to be termed as "best available techniques" (Feenstra et al. 2009).

Based on "semi-quantitative" ranking methodology the ScorePP project (Lecloux 2010) concluded, that the substitution of NP and NPE is technically feasible and regulations strongly limit the use of these substances. Thus, actions should be focused on effective substitution and on strict application of existing regulations. It is however important to investigate if the substitutes pose an unacceptable burden to the health and the environ-ment.

Based on a rather qualitative scoring methodology the SOCOPSE project concluded that the substitution of NP and NPE is one of the advisable abatement measures for reducing NP emissions to water (Feenstra et al. 2009).

6.2.3 Costs

- Estimated substitution costs reported by an English company are:
- Substitute costs 10% more per unit than the NP based additive, increasing the total product cost by less than 1%
- a one-off cost of £7 million is expected for the reformulation and changing of materials for decorative emulsions, with these relating to per unit costs of £ 0.10/litre of product (EUR 0.1 / litre of product with exchange rate EUR 1 ≈ £ 0.9, July 2011)

- It will take 2-7 years to fully substitute NPE in paint products
- Costs of developing substitutes for the other applications are currently unknown, although these are expected to be significantly higher than those quoted above in the case of waterborne cleaners (on a per unit basis).

In contrast, another English company indicated that substitution of alcohol ethoxylates is roughly 50% more costly, with this implying a 2.5% increase in end-product costs (RPA 1999).

6.2.4 Cost-Effectiveness Analysis

The used amounts in manufacture of paints in Finland are 2 tons NP/year and 154 tons NPE / year (Mehtonen 2011).

Emissions to waste water from manufacture of paints in Finland: 780 kg NP&NPE /year (Mehtonen 2011). Emissions from application of paints in Finland are assumed to be minor compared to emissions from manufacture of paints (approximately 5%, Mehtonen 2011).

Emissions to surface water via waste water treatment plant effluents due to manufacture of paints in Finland: 39-78 kg NP&NPE /year with assumption of 90-95% reduction efficiency in waste water treatment plant.

Finland, Cost Scenario P1:

The total one-off investment costs to substitute NPE with alcohol ethoxylates in all paint product types in Finland are estimated to be EUR 1 215 000 with the following assumptions;

- One-off unit costs of EUR 0.16 / litre of product (discounted with annual inflation rate 3% from year 1997 to 2011) is expected for the reformulation and changing of materials for all paint product types
- No additional operational and maintenance costs (i.e. assumed to be same as for NP/NPE)
- No end of life costs
- Used amount of NP/NPE in manufacture of paints in Finland is 156 tons NPE / year

- The total amount of paint formulation containing NPE manufactured in Finland is approximately 7 800 000 l product assuming that
- 100% of total amount manufactured paint containing NPE is decorative emulsion type (i.e. no industrial use at all)
- The typical paint formulation contains 2% NPE
- 1 kg paint end product ≈ 1 l paint end product
    - It will take 5 years to fully substitute NPE in paint products
    - Time span for the analysis is 25 years

Cost Scenario	Costs over 25 a [MEUR]	Emission reduction to surface water [kg/25 a]		Cost Effectiveness [EUR/kg]	
		Min.	max.	max.	min.
P1	1.22	780	1560	1564	782

6.3 Measure 2: Ban of NP and NPE in textiles

6.3.1 Description of source

Nonylphenol ethoxylates are used in several processes of textile manufacture including scouring, fibre lubrication and dye levelling. The main use is in wool scouring where natural oils are removed from the wool. NPEs are used because of their detergent and fibre lubricating (conditioning) properties and because they are not adsorbed into the wool (EU RAR 2002). It seems that NPE may be present in both fabric (Seppänen 2010) and printing sections of textiles (Pedersen & Hartmann 2004).

Since 2005 the use of nonylphenol and nonylphenol ethoxylates in textile industries is prohibited in the EU due to Directive 2003/53/EC. However, the use is still allowed in closed systems, and there is a possibility that NPs and NPEs can remain in textiles and leather from these processes. Concerning the Baltic Sea catchment area, NP and NPE are used in manufacture of textiles at least in Estonia and Germany (Mehtonen 2011).

Nevertheless, manufacturing of textiles in the EU has decreased over the years largely due to cheaper manufacturing costs in other countries. Many textiles are imported to the EU from non-European countries, such as China, India and Bangladesh (developing economies). Thus, emissions of NP and NPE are possible e.g. via washing of textiles and via leaching from landfills containing textile and other articles containing NP or NPE. The emissions from washing of textiles imported outside EU are estimated to be much more significant than emissions from landfills. The washing of textiles containing NPE is estimated to be one of the main emission sources in many Baltic Sea countries (Månsson 2009, Mehtonen 2011 & COHIBA WP4 data).

Emissions of NP and NPE from washing of textile articles are mainly lead to effective municipal waste water treatment plants, but from scattered settlements emissions are emitted to environment via smaller and less effective waste water treatment plants.

6.3.2 Description of measure

Potential measures can be divided in two categories:

1.) Measures reducing emissions from the use of NPE in textile industry at EU level (point sources) and respective emissions via washing of textile articles (diffuse emissions)

2.) Measures reducing the emissions via washing of textile articles imported from outside the EU (diffuse emissions).

Total ban of NP and NPE use in textile industry on an EU level would facilitate the authority control of NPE use in the textile industry. Currently for example, the national registers of chemical products (existing at least in Denmark, Finland & Sweden) indicate if NPE is used or is not used in country, but it does not give information on locations where NPE is used (i.e. "downstream users") or if NPE is actually used in processes with no release into waste water.

The ban of NP and NPE in textile articles imported from outside the EU would allow the use of different authority control procedures concerning textiles on the market and textiles imported from outside the EU:

Textiles on the market:

The competent authority (e.g. in Finland the Finnish Safety and Chemicals Agency, Tukes) conducts surveillance of consumer articles, including textiles, on the market. The aim of the surveillance is to make sure that the articles fulfil the requirements of legislation and do not pose a risk to the health of the consumer. According to the General Product Safety Directive (GPSD) implemented by the Act on safety of consumer products and services (75/2004), the main responsibility for the safety of the article lies with the producer, importer or supplier of the article. The market surveillance is conducted preventively and reactively, including e.g. spot checks, notifications and surveillance projects. Spot checks include analysing articles purchased from stores. Notifications on dangerous articles include those made by consumers, companies and other authorities (e.g. RAPEX). Surveys are used to find out the compliance with legislation by article groups (Assmuth et al. 2011).

If the article is found not to comply with legislation, the marketing of the article is prevented, the article is withdrawn from the market or the article is collected back from the consumers. The authorities can prohibit the sale or production of the article, as well as the export. The article can be ordered to be destroyed or the company can repair or replace the article with safer one (Assmuth et al. 2011).

Textiles imported from outside the EU:

Customs services conduct surveillance of consumer products, including textiles, coming from outside the EU area. The surveillance is based on the product safety legislation (75/2004) and the food legislation (23/2006). Customs also conduct surveillance of certain articles based on the Annex XVII of the REACH Regulation. These include e.g. cadmium in certain plastics and paint, migration of nickel from articles in contact with the skin, certain azo dyes in textile and leather articles in contact with the skin and certain phthalates in toys and childcare articles. The selection of these articles is based on legislation given on the basis of the product safety legislation. The tested articles and chemicals are mainly those regulated by national and/or EU legislation (Assmuth et al. 2011).

If Customs find articles not complying with legislation, the importer has to destroy the imported batch, apply for a licence to export the batch from the country or change the article to comply with legislation (e.g. on labelling). The authorities can also write a notice to the importer, if the limit value of the chemical has only been slightly exceeded. In case of articles coming inside the EU area, the company can voluntarily withdraw the article from the market or the authorities can order the article to be withdrawn from the market and the consumers (Assmuth et al. 2011).

Currently these authority control procedures are not possible because the import of NP/NPE in textile articles outside the EU is legal. Thus, the total ban of NP and NPE use in the textile industry in the EU area would clearly facilitate authority control on NPE use in the textile industry because a total ban is easier to control than a conditional restric-tion/ban. This measure would, if not totally end, at least decrease the emissions from textile manufacture and to some extent decrease emissions via washing of textiles. Additionally, the ban on NP/NPE in textiles imported from outside the EU would effectively decrease the emissions via textile articles (especially via washing of textiles) if authority control is effective enough at market level as well as at the European borders by Customs.

6.3.3 *Cost-Effectiveness Analysis*

The cost-effectiveness analysis (CEA) for textile processing (use allowed if no emissions) is not performed in this study. Nevertheless, it can be shortly mentioned that in the textile sector, (wool scouring) with careful detergent selection and process optimisation, substitution of NPEs should not result in increased costs to wool scourers. It was revealed that while alcohol ethoxylates were around 20% more expensive than alkylphenol ethoxylates (APE) such as nonylphenol ethoxylates, the cost of scouring of raw wool could be reduced significantly with alcohol ethoxylates. This saving is due to increases in process efficiency which reduced detergent use from 0.8% to 0.5% (RPA 1999). It is not possible to estimate the costs for the emission source of textiles articles concerning measures on both textile use in the EU and import of textiles outside the EU.

6.4 Measure 3a: Advanced waste water treatment – AC treatment

6.4.1 *Description of source*

Input of NP and NPE to municipal waste water treatment plants depend on local conditions of the urban area e.g. pattern of indirect dischargers (industrial sites), use pattern of different kind of products, user behaviour, pollutant load from urban run-off (roofs, streets etc.) in the case of combined sewer systems. On industrial sites, loads of different substances to industrial treatment plants depends strongly on the branch of industry and their use of different chemicals. The load of pollutants in waste water varies in different cities and can also vary in time. Therefore predicting the load of possible pollutants at MWWTPs has a high uncertainty.

Nonylphenol removal in conventional waste water treatment (mechanical, biological and chemical treatment and improved nitrogen removal) has proved to be efficient. Since NP has a low solubility, high sorption potential and low volatilisation potential, degradation and sorption in sewage sludge are the main mechanisms involved in the removal of NPs from waste waters. The removal efficiencies from the water phase in WWTP is higher than 95%, although more than 75 % of the total concentration in influents is transferred to the sludge (Seriki et al 2008).

6.4.2 *Description of measure*

Activated Carbon Filters (AC treatment) for removal of pollutants from waste water are a proven technology. AC filters have a large surface area and functions as an effective sorbent for many substances. Different technical systems are commercially available (e.g. powder (PAC) and granular activated carbon (GAC)).

6.4.3 *Effectiveness*

Effectiveness of AC filters at WWTP depends on the concentration range of pollutants, technical parameters and matrix. At well maintained MWWTP reduction rates for NP of 50% to 99% can be observed. In Danish COHIBA case studies the reduction efficiency reached only 25%. Some of the pollutants cannot be removed from the activated carbon and incineration is necessary (incineration is not included in the calculations). Additionally, effectiveness depends on matrix (for example COD other micropollutants etc.) and concentration of sorbent. It has been observed that NPs at lower concentrations have a lower adsorption capacity to AC filter than at higher concentrations. Also the amount of natural organic carbon can reduce to some extent the adsorption on NP to AC (Yu Zirui et al. 2008).

6.4.4 *Secondary environmental effects*

AC filters at WWTPs also affect emission of several other hazardous substances of special concern to the Baltic Sea, which are typically present in municipal waste waters in very low concentrations. AC has large cross substance effects e.g. elimination of TBT, PFOS, NP, Cd, Hg and other pollutants. In a similar way AC treatment affects pollutants in industrial waste waters. Depending on the branch of industry, removal rates can be significantly higher. AC filters have also potentially large positive water related secondary environmental effects on the reduction of phosphate emissions and other pollutants, such as heavy metals, organic micropollutants, pharmaceuticals and their metabolites or endocrine disrupters. Negative secondary environmental effects are related to energy use and emissions during the construction and operation and for manufacture of activated carbon (Marscheider-Weidemann 2011).

6.4.5 Technical feasibility

AC filters are a proven technology.

6.4.6 Secondary socio-economic effects (including indirect costs)

If large MWWTPs are equipped with AC treatment, the question of financing has to be considered. One option is that the respective MWWTPs charge additional costs to their clients. Large MWWTPs often show lower costs per capita than smaller MWWTPs. The other option is to have the costs paid by all citizens (e.g. via taxes), as the whole community benefits from the removal of pollutants. The latter option was put into practice in Switzerland (total costs of waste water treatment rose by 6%). A very good, but hard to put into practice option is to tax production and use of products containing hazardous substances, which would follow the polluter pays principle.

6.4.7 Geographical and time scale of effects

This measure becomes effective immediately and has a technical life span of 80 years. It seems sensible to target large plants because of economy of scale effects. Concerning the Baltic Sea catchment area, MWWTPs near the shore can be interesting targets (Table 2).

6.4.8 Cost-Effectiveness Analysis

Based on the data presented in the proceeding chapter, two scenarios for Cost-Effectiveness Analysis (CEA) are derived.

Table 2: Best case and worst case scenario for CEA of AC treatment

Best case scenario	High load of source	High effectiveness of measure	Low costs of measures
	56 kg/a	99%	EUR 0.02-2.1 / p.e. (EUR 0.002-0.02 / p.e. SOCOPSE)
Worst case scenario	Low load of source	Low effectiveness of measure	High costs of measure
	19 kg/a	25%	EUR 0.2- 24.7 /p.e. (EUR 0.26-2.6 / p.e. SOCOPSE)

The presented calculations of CE are oversimplified examples to illustrate certain important aspects, like the strong dependency of CE from actual load of the source. This CE calculation example is based on measured average values from WWTP data from COHIBA Work Package 3.

- Measured lowest average effluent concentration of NP is 0.2 µg/l (SOCOPSE 1.0 µg/l)
- Measured highest average effluent concentration of NP is 0.6 µg/l (SOCOPSE 10.0 µg/l)
- Load is 19–56 kg/a (at a 780 000 p.e. WWTP with an assumed flow of 326 l/d p.e.)
- Effectiveness assumed with 25–99 %
- Costs assumed with 0.02–24.7 EUR /p.e. (EUR 0.002-2.6 /p.e. SOCOPSE)

Worst case scenario: CE = EUR 140 000–19 million /kg (EUR 200 000–2 million /kg SOCOPSE)

Best case scenario: CE = EUR 12 000–1.6 million /kg (EUR 1 700–17 000 /kg SOCOPSE)

According to another Danish case study the calculated cost-effectiveness of AC filters for the removal of NP and NPEs is EUR 5 million /kg with an effectiveness of 25%.

6.5 Measure 3b: Advanced waste water treatment - membrane filtration

6.5.1 Description of source

For the description of source please refer to Chapter 6.4.1

6.5.2 Description of measure

Membrane filtration includes Nanofiltration (NF) or Reverse Osmosis (RO). NF is applied to remove larger organic molecules and multivalent ions in order to recycle and reuse the waste water or reduce its volume and simultaneously increase the concentration of contaminants to such an extent that subsequent destruction processes are feasible. RO is a process to separate dissolved constituents from the water phase down to virtually all organic and ionic species. It is applied when a high purity grade of water is required. The recovered water phase is usually recycled and reused.

6.5.3 Effectiveness

Effectiveness of NF filters removing NP from MWWTP effluents is found to vary from 70% to 100%. In the case of RO the effectiveness is generally above 98% (Feenstra et al. 2009).

6.5.4 Secondary environmental effects

NF and RO filtration also affects emissions of other hazardous substances of special concern to the Baltic Sea. These measures have also potentially large positive water related secondary environmental effects on other pollutants, such as heavy metals, organic micropollutants, pharmaceuticals and their metabolites or endocrine disrupters.

6.5.5 Technical feasibility

The molecular weight cut-off (the molar mass above more that 90% of the compounds are rejected) of present nanofiltration membranes is situated in the range of 200-500 g/mol, which corresponds to the molar mass of several organic micropollutants in surface and waste water. The polarity is also an important parameter to describe rejection; the polarity is related to the octanol/water partition coefficient. Negatively charged solutes are better rejected due to the negative charge of the membrane surface of common nanofiltration membranes at neutral pH-conditions (Feenstra et al. 2009).

6.5.6 Cost-Effectiveness Analysis

Based on the data presented in the proceeding chapter, two scenarios for cost-effectiveness analysis (CEA) are derived (Table 3).

Table 3: Best case and worst case scenario for CEA of membrane filtration

Best case scenario	High load of source	High effectiveness of measure	Low costs of measures
	56 kg/a	100%	EUR 0.1-1.3 / p.e.
Worst case scenario	Low load of source	Low effectiveness of measure	High costs of measure
	19 kg/a	70%	EUR 0.5- 5.4 / p.e.

The presented calculations of CE are oversimplified examples to illustrate certain impor-tant aspects, like strong dependency of CE from actual load of the source. This CE calculation example is based on measured average values from WWTP data from COHIBA Work package 3.

- Measured lowest average effluent concentration of NP is 0.2 µg/l
- Measured highest average effluent concentration of NP is 0.6 µg/l

Load is 19– 56 kg/a (at a 780 000 p.e. WWTP with an assumed flow of 326 l/d p.e.)

- Effectiveness is assumed with 70 - 100 %
- Costs are assumed with EUR 0.1 – 5.4 /p.e.

Worst case scenario: CE = EUR 400 000 – 4.2 million /kg

Best case scenario: CE = EUR 93 000 – 990 000 /kg

6.6 Measure 3c: Advanced waste water treatment – oxidative techniques

6.6.1 Description of source

For the description of source please refer to Chapter 6.4.1

6.6.2 Description of measure

Oxidative techniques have been used for the treatment of waste water containing recalcitrant organic compounds such as pesticides, surfactants, colouring matters, pharmaceuticals and endocrine disrupting chemicals. Moreover, they have been successfully used as pre-treatment methods in order to reduce concentrations of toxic organic compounds that inhibit biological waste water treatment processes.

Oxidative techniques “crack” organic compounds with the aid of strong oxidants such as ozone or hydrogen peroxide. The treatment principle involves a non-specific or specific reaction of the oxidant with the organic compounds, which are oxidised and degraded to smaller molecules. The extent to which the process proceeds depends directly on the nature of the organic compounds, the nature and the concentration of the dosed oxidant and the contact time. Advanced Oxidation Processes (AOP) combine different oxidation techniques. Free radicals can be formed which make the process several thousand times faster. Applicable combinations are ozone/hydrogen peroxide, ozone/UV and hydrogen peroxide/UV. In the case of high hydrogen peroxide (H2O2) dosage, high UV intensities are however required. Another disadvantage is the formation of new toxic compounds.

6.6.3 Technical feasibility

Oxidative techniques are potentially applicable for the removal of endocrine-disrupting chemicals (EDCS) like nonylphenol (NP).

A particular point of importance is the required UV-dosage in relation to the transmission (turbidity) of waste water, which is negatively influenced by dissolved and suspended organic material. Removal of these components in an effective pre-treatment step is crucial.

6.6.4 Effectiveness

According to SOCOPSE the chemical oxidation treatment efficiency for NP removal is 90%.

6.6.5 Cost-Effectiveness Analysis

Based on the data presented in the proceeding chapter, two scenarios for cost-effectiveness analysis (CEA) are derived (Table 4). The treatment efficiency is 90% in both cases (SOCOPSE).

Table 4: Best case and worst case scenario for CEA for oxidative techniques

Best case scenario	High load of source	High effectiveness of measure	Low costs of measures
	56 kg/a	90%	EUR 0.2-4.9 / p.e.
Worst case scenario	Low load of source	Low effectiveness of measure	High costs of measure
	19 kg/a	90%	EUR 0.5-14.8 / p.e.

The presented calculations of CE are oversimplified examples to illustrate certain impor-tant aspects, like strong dependency of CE from actual load of the source. This CE calculation example is based on measured average values from WWTP data from COHIBA Work package 3.

- Measured lowest average effluent concentration of NP is 0.2 µg/l
- Measured highest average effluent concentration of NP is 0.6 µg/l

Load is 19– 56 kg/a (at a 780 000 p.e. WWTP with an assumed flow of 326 l/d p.e.)

- Effectiveness is assumed with 90 %
- Costs are assumed with EUR 0.2 – 15 /p.e.

Worst case scenario: CE = EUR 360 000 – 1.2 million /kg

Best case scenario: CE = EUR 120 000 - 3.9 million /kg

6.7 Measure 4: Sludge treatment – anaerobic and aerobic digestion

6.7.1 Description of source

Nonylphenol removal in conventional waste water treatment, including mechanical, biological and chemical treatment and improved nitrogen removal, has proved to be efficient. As NP has a low solubility, high sorption potential and low volatilisation potential, degradation and sorption in sewage sludge are the main mechanisms for the removal of NPs from waste waters and more than 75 % of total concentration in influents is transferred to the sludge (Seriki et al 2008).

There are three major sludge disposal pathways for municipal sewage sludge. Sludge can be either used for application to agricultural land, or can be incinerated, or can be landfilled. To reduce emissions of NP and NPE, landfill leachate can be collected and treated in WWTPs. In this case, the same reduction of NP and NPE as for mechanical, chemical and biological treatment in WWTPs can be assumed.

In some cases sludge can be incorporated in cement production. Previously, disposal to the coastal waters (sea dumping) was also widely practiced, however this practice is now banned within the EU since 1998 (Donner et al. 2010). The yearly amount of sewage sludge generated in Finland is ca. 1.1 – 1.2 million tonnes.

6.7.2 Description of measures

According to some Finnish studies composting of sludge reduced the amount of NPs by 51%. After sludge application to soil NP observed to degrade by 92% during three months. According to another study, NP removal efficiency was 100% with aerobic digestion. The only disadvantage mentioned for this process was the retention time of 20 days (Donner et al. 2010). Good NP degradation results were achieved by using firstly anaerobic digestion and subsequently post aerobic treatment (removal efficiency 97.8 %). Based on these studies, NPs are readily biodegradable under aerobic conditions.

Unlike NPs, NPEs are found to degrade also in anaerobic conditions. Rotting of sludge degrades NPE to NP and polyglycol. Polyglycol continues to degrade under anaerobic conditions but NP accumulates to sludge (Itävaara et al. 2007).

Due to the degradation potential of both NPs and NPEs , an effective measure for the reduction of these substances from sludge could be anaerobic treatment followed by aerobic treatment.

6.7.3 Cost-Effectiveness Analysis

Concerning composting in order to have some kind of overview of sludge handling expenses, the yearly costs for plants handling 5 000 t/a of sewage sludge are EUR 80 /t (according to the report of Finnish Ministry of Agriculture and Forestry). In plants treating sludge amounts of 75 000 t/a or more, costs per unit are EUR 71 /t. Generally, smaller plants have higher costs per unit because of higher relative investment costs compared to larger plants (Pöyry Environment Oy 2007).

The estimated amount of sludge formed yearly in Finland is about 1.2 million tonnes. The yearly amount of NP entering sewage sludge, based on COHIBA WP3 results, is about 11 000 kg. It has been observed that the reduction potential of NP in composting is 51%. Costs for NP reduction from sewage sludge by composting is ca. EUR 16 100 per reduced kg.

Concerning rotting and composting, a report of Finnish Ministry of Agriculture and Forestry indicates yearly costs of EUR 140 /t for plants handling 20 000 t/a of sewage sludge. In plants treating sludge amounts of 75 000 t/a or more, costs per unit are EUR 120 /t (Pöyry Environment Oy 2007). However, in these calculations, NPs and NPEs are not segregated from total costs.

6.8 Measure 5: Urban run-off management

6.8.1 Description of source

NP and NPE input from urban run-off can be a direct source to the Baltic Sea coastal areas. COHIBA sampling results show average concentrations of 1.9 µg/l for NPs and 1.11 µg/l for NPEs in storm waters. The overall highest observed maximum NP concentration in COHIBA sampling was 15 µg/l. The average annual (AA-EQS) and maximum allowable concentration (MAC-EQS) for 4-nonylphenol in inland and other surface waters are 0.3 µg/l and 2 µg/l, respectively. As can be seen, both of these values were exceeded in COHIBA sampling.

6.8.2 Description of measures

According to a Danish COHIBA case study, urban run-off management is a potential way to reduce NP/NPE load to the Baltic Sea.

6.8.3 Effectiveness

According to Danish COHIBA studies, over 40% reduction of total NP and NPE load to the Baltic Sea can be achieved by urban run-off management by using double porous filters (DPF) as treatment technology. Another, even more effective way to reduce the amounts of NP/NPEs from storm waters is filtration. For example sand filtration can reduce 50-80 % and infiltration basins 90% of NP/NPEs from storm waters (ScorePP, Source Control Options for Reducing Emissions of Priority Pollutants). According to the ScorePP–project, constructed wetlands are also effective in reducing NPs loads from storm waters, showing removal rates of 90%.

6.8.4 Secondary environmental effects

Urban run-off management has several other advantages: by using filtration systems or constructed wetlands other harmful substances, such as nutrients and solid substances can be reduced efficiently. According to COHIBA studies, sand filtration can reduce the amount of solid substances by 70-99 % and the amount of mercury by 13%.

6.8.5 Cost-Effectiveness Analysis

According to Danish COHIBA studies, double porous filters (DPF) for removal of NP and NPEs from storm water cost EUR 0.6 million /kg. However, this measure can only be implemented for small hydraulic loads (< 20m3/h). Concerning sand filtration for flows of 7500 m3/h, investment costs can be assumed to be about EUR 8.1 million and operational costs EUR 0.8 million /year.

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VII. Octylphenol (OP) and octylphenol ethoxylates (OPE)

7.1 Measure 1: Substitution of OPE in textile printing

7.1.1 Description of source

OPEs are used in the textile industry for printing purposes like e.g. T-shirt printing. They are used in specific emulsion processes as glues, binders, emulsifiers for dyestuffs, emulsion polymerisation, etc. Emulsion systems are based on polyvinyl chloride or polyurethane. Most of the OPE is physically bound in the polymer matrix which sticks to the textile. However, although this is a legal use in water-free printing processes, a lot of the OPE, and NPE as well, is washed out over the lifetime of the textiles and ends up in waste-water.

There are many different chemical companies who offer OPE containing products such as Dow (Triton X series) or Huntsman (e.g. in the Surfonic®OP, Empilan OP and Teric® X series).

7.1.2 Description of measure

With regard to non-ionic surfactants, replacing OPEs with other materials appears to be actively promoted by the chemical companies. Several alternatives to OPEs are available. Typical substitutes are fatty alcohol ethoxylates, for more difficult applications other branched alcohol compounds like Guerbet alcohol are used. These substitutes can cost twice as much as OPEs. Other identified alternatives to OPEs are based on 4-tert-pentylphenol or dodecylphenol . Potential risks have been identified for both.

Recent work undertaken under the OECD SIDS programme indicates that while some fatty alcohols may be toxic to aquatic organisms, they do not bioaccumulate or have endocrine disrupting effects and are not expected to be as persistent as OP (personal communication, 2006)

7.1.3 Effectiveness

The effectiveness of the measure is high, although it will take some years before a substitution process is fully completed. RPA gives a time span of 3 years as an example for one company (RPA, 2006).

7.1.4 Costs

The average price for alkoxylates in 2006 was estimated to be ca. 1,100 € per tonne (Frost&Sullivan, 2007).

Typical substitutes are fatty alcohol ethoxylates which are ca. 10-20% more expensive than OPE. Branched alcohol compounds like Guerbet alcohol, cost approximately twice as much as the currently used OPE compound.

The costs for the chemicals are only one part of the substitution costs (RPA, 2006). Reformulation of the products to prevent worsening in performance is often necessary, as well as plant trials, customer trials and support. RPA (2006) estimates the costs for a substitution programme at around € 58,000 (£ 50,000) in technical support, sampling and staff time.

7.1.5 Secondary environmental effects

As with all substitutions, care must be taken that the substitute chemicals are not dangerous for the environment.

7.1.6 Technical feasibility

In general, substitution of OPE in textile printing is technical feasible. Because of the good performance of OPE, it might be necessary to change the formulation and use more than one chemical to substitute OPE.

7.1.7 Secondary socio-economic effects (including indirect costs)

Because available substitutes cost only slightly more, a significant increase in product costs is unlikely.

7.1.8 Geographical and time scale of effects

Most textiles are printed outside Europe and are then imported. Hence, along with substitution in Europe, an import control of textiles containing OPE must also be put in place.

7.1.9 Political enforceability

There is no political reason standing against the introduction of this measure.

7.1.10 Cost-effectiveness analysis

The costs for substitutes are somewhere between 120 to 200 % of the OP/OPE costs. Those are pure chemical costs, without expenses for reformulation. Based on the costs discussed in Chapter 5.4 the substitutes have prices between 1.20 and 2.20 Euro per kg (SASOL, 2011).

7.2 Measure 2: Waste management – controlled incineration of waste tyres

7.2.1 Description of source

OP-based resins are used to increase the tackiness of rubber in tyres and improve adhesion of the different layers during vulcanisation. The resins are usually added to rubber in amounts of up to 1.5% of the rubber formulation although the maximum figure for the percentage of resin in rubber used in tyres could be as high as 10% (EA, 2005). In 2010, 4.5 million tonnes of tyres were produced in 91 tyre manufacturing facilities across Europe (ETRMA, 2011).

According RPA (2008) the primary function of the OP formaldehyde resin is to maintain the internal structural integrity of a so called “green tyre” prior to curing by high temperature vulcanisation. According to the tyre industry, no OP is released to the environment during the tyres’ service life (ETRMA, 2007). The release or no release of OP from tyres is a field for further research. Nevertheless, it is documented in literature that, if the tyres after service life are shred, these grounded tyre material release OP into the environment.

In some studies only low OP concentrations were found in leachate from products from end of life tyres, like e. g. 0.1 µg/l from light fill noise barriers (Aabøe et al., 2004). Other studies reports higher concentrations, like 3.6 µg/l (Niva, 2005) from artificial turf systems or 3.6 mg/l from rubber granulate (NBRI, 2004), so an environmental effect can be feared.

7.2.2 Description of measure

The substitution of OPE resins in tyres seems to be difficult. Thus, it is proposed to collect all waste tyres and reuse them as whole tyre if possible without destroying the matrix. After their service life, the tyres should be oxidized in steel mills, incineration plants, foundries or cement kilns. However, they should not be grinded and used in civil engineering, where significant amounts of OP can be set free.

7.2.3 Effectiveness

The effectiveness is high after implementation of the measure.

7.2.4 Costs

In collection systems operated by producers, these companies or importers of tyres charge a fee (e. g. 57 EURO/t in Estonia) for collection and disposal of end of life tyres. The disposal fee per tonne of tyres for passenger cars is around 65 € in Germany. On the other hand, the waste industry sells the tyres to industries which use their energy content.

This measure would not increase current end of life costs for tyres in the Baltic Sea countries (BSC).

7.2.5 Secondary environmental effects

The emission of high amounts of zinc can also be stopped by this measure. Zinc oxide is used in rubber industry as activator for sulphur vulcanisation. The concentration in tyres is 1 to 2 %.

Several Life Cycle Assessments (LCAs) in the literature show that the recycling of used tyres saves more energy and is environmentally better than incineration in cement kilns, if the recycled material from tyres substitute virgin rubber (Schmidt et al., 2009).

7.2.6 Technical feasibility

The incineration of tyres is already common practice in most countries in the BSC area.

7.2.7 Secondary socio-economic effects (including indirect costs)

Artificial turf systems with grinded rubber from disposed tyres are used for baseball fields in the US. They are not used widely in the BSC.

By the thermal use of end of life tyres in cement kilns etc. other energy sources could be saved.

7.2.8 Geographical and time scale of effects

In some countries capacities for using tyres for energy recovery have to be built up.

7.2.9 Political enforceability

In the BSC producer operated collection systems already exist in Sweden, Finland, Estonia and Poland. Lithuania and Latvia are in the process of setting up such systems, because Landfill Directive 1999/31/EC prohibits landfilling of used tyres. In Germany there is a free market system, in which owners or garages have to pay a fee to get rid of old tyres.

7.2.10 Cost-effectiveness analysis

This measure is supposed to be cost-neutral. The amount of reduced emissions cannot be calculated, so cost-effectiveness can’t be calculated.

7.3 Measure 3: Voluntary agreement to stop using OP

A voluntary reduction agreement was developed in the UK in anticipation of the then impending EU-wide Directive 2003/53/EC concerning the marketing and use of NP and NPE. The UK industry agreed not to promote OPEs as substitutes for NPE. End users undertook to stop using NP/E and OP/E in new formulations from Jan 2004 and to phase out their use in existing formulations as soon as possible or by Dec 2004 at the latest.

7.3.1 Description of source

Similar to the use of NP and NPE before EU Directive 2003/53/EC, there are a lot of different possible uses for OP and OPE. OP is also an impurity of NP formulations, which are still widely used in metalworking, in the manufacture of coatings, textiles and leathers, and in certain areas not covered by the EU ban such as emulsion polymerization and chemical intermediates in manufacturing processes. These applications are major emission sources.

7.3.2 Description of measure

The UK's voluntary agreement, which comprises separate measures for suppliers and downstream users, should be applied to the BSC. Under the suppliers' agreement, manufacturers and suppliers would take action to support risk reduction by promoting safer substitutes to NPEs (such as the similar alcohol ethoxylates), and would inform and advise their customers about the voluntary agreement.

As in the UK, the following producers, trade associations and downstream users should be part of this agreement (DEFRA, 2004):

- Chemicals supply industry
- Cosmetic, toiletry & perfumery industry
- Specialty chemicals industry
- Paper industries
- Cleaning products industry
- Crop protection industry
- Fragrance industry

7.3.3 Effectiveness

The effectiveness of this measure is high, since the involvement of importers and producers opens up the possibility to abandon all uses of OP/OPE. Because of the desired use of OP in tyres (see above) a ban on OP/OPE is difficult to implement.

7.3.4 Costs

The cost to UK industry of one-time reformulation and changes in raw materials costs due to the use of alternatives to NP/NPE and OP/OPE was estimated by DEFRA at about 6.4 million € (£4.3 million; Edser, 2004).

Today, the cost will be significant lower because many companies who are big players around the BSC produce in the UK and are already parties to the UK voluntary agreement (including Akzo Nobel Surface Chemistry, BASF, Croda Chemicals Europe, Dow Europe, Goldschmidt UK, Huntsman Surface Sciences UK, Shell Chemicals, Stepan UK, Uniqema). They have already invested in R&D of environmentally friendly, but functionally equivalent, alternatives to NPE and OPE (Edser, 2004).

7.3.5 Secondary environmental effects

There are no expected secondary effects.

7.3.6 Technical feasibility

Measure is already established in UK.

7.3.7 Secondary socio-economic effects (including indirect costs)

The associations and companies who sign the voluntary agreement can show that they are first movers in environmental protection and use it in their press work.

7.3.8 Geographical and time scale of effects

Voluntary economic instruments like voluntary agreements do not seem to be working in Russia at all, mainly because of the low level of general public awareness and a lack of demand for cleaner or safer products (Toropovs, 2011).

7.3.9 Political enforceability

Voluntary agreements are normally used on a country scale. It might be a challenge to implement them on BSC level.

7.3.10 Cost-effectiveness analysis

The costs of substitutes are between 120 and 200 % of the OP/OPE costs. Those are pure chemical costs, without expenses for reformulation and a mechanism for controlling the agreement. The substitutes have prices between 1.20 and 2.20 Euro per kg.

7.4 Measure 4: Advanced waste water treatment – AC treatment

7.4.1 Description of source

Pollutant loads in waste water vary greatly in different cities/districts/MWWTPs (e.g. depending on indirect industrial dischargers) and can also vary markedly in time. Therefore, predicting the kinds and loads of pollutants at MWWTPs has a very high uncertainty.

Primary degradation of OPEs through ether cleavage in wastewater treatment plants generates more persistent shorter chained OPEs and octylphenol (Giger et al., 1984). The elimination rates for OP e.g. in German MWWTP are between 73 and 98 % (great influence of the limit value; MUNLV, 2004). The main treatment plant in Vienna has a rate of 50 % (Scharf et al., 1998). Rates ranging from 7 to 100 % were found in 16 Canadian plants (Bennie et al., 1998).

In the figure below, concentrations of OP in MWWTP effluent in different countries are presented.

Figure 3: Concentrations and loads of OP in MWWTPs as found in a literature review

Load:

For OP

between 6 – 36 µg/ day per capita (measured data from COHIBA-WP3)
Concentrations [µg/l]:
Canada: 0.12– 1.7 (mean0.69) (Lee et al.,1995)
Japan: 0.02– 0.48 (Isobe et al., 2001)
USA : 0.15 (Rudel et al., 1998)
< LOD– 0.673 (0.072) (Snyder et al., 1999)
0.21-1.58 (mean, summer/winter; Loyo-Rosales, 2007)
Germany : < LOD– 0.073 (0.014) (Kuch et al., 2001)

Range: assumed 200 l per capita, 4 to 340 µg/day per capita, equals ~1.4 to 124 mg/a per capita

7.4.2 Description of measure

Activated carbon (AC) filters for removal of pollutants from wastewater are a proven technology. AC has a large surface area and is an effective adsorbent for many substances. Different technical systems are commercially available (e.g. powder (PAC) and granular activated carbon (GAC)). These adsorbents can be used in different processes in the MWWTP, e.g. as fixed bed after the third treatment stage or mixed with the waste water.

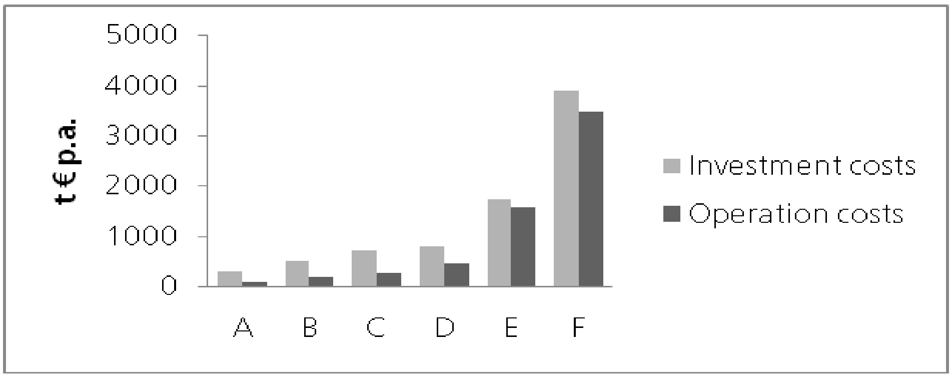
7.4.3 Effectiveness

The effectiveness of AC filters in removing OP at MWWTP depends on the concentration range of pollutant, technical parameters and the matrix. At well maintained MWWTPs reduction rates of 75 % in the effluent of MWWTPs using PAC can be observed (Fahlenkamp et al., 2008).

7.4.4 Costs

Economic analysis from the project “StrategyMicroPoll” (Switzerland, Ort et al., 2009) found costs of 10-60 € per person and year, including discounted investment costs and running costs, see Figure 5. Specific costs are strongly dependent on the size of the MWWTP (large WWTP, low specific costs, economy of scales), see data for 6 plants in Figure 4.

Figure 4: Yearly investment costs and running costs, economic data from StrategyMicroPoll



7.4.5 Secondary environmental effects

AC filtration at MWWTPs, sometimes called the 4th stage of waste water treatment, affects emissions of the other of the 11 HS of special concern to the Baltic Sea, which are typically present in municipal waste water in very low concentrations. Filters also have other water related secondary environmental effects: on emissions of phosphate, heavy metals, organic micro pollutants (which are not on the HELCOM list), pharmaceuticals and their metabolites, or endocrine disrupters.

Additionally, AC treatment has significant cross substance effects and enables a simultaneous elimination of other hazardous substances, such as TBT, PFOS, nonylphenol, Cd, and Hg. This makes it possible to calculate effectiveness rates and loads for all 11 HS treated at MWWTPs.

Negative secondary environmental effects are related to e.g. higher energy use combined with greenhouse gas (GHG) emissions during construction and operation, and for manufacture of activated carbon (Wenzel, 2008).

7.4.6 Technical feasibility

AC treatment is an available technology, a well-functioning MWWT with low concentrations of suspended solids and dissolved organics is necessary.

7.4.7 Secondary socio-economic effects (including indirect costs)

The costs of advanced waste water treatment can be passed on to the customers of the MWWTP. As shown above, large MWWTPs normally have lower per capita costs than smaller MWWTPs.

The other option is to have the costs paid for by all citizens via taxes, as the whole community benefits from a healthier environment free of toxic substances. The latter option was put into practice in Switzerland (total costs of waste water treatment rose by 6%).

7.4.8 Geographical and time scale of effects

Most of the emissions from urban stock are channelled through MWWTPs, urban runoff and landfills. The types and loads of pollutants in waste water are dependent on local conditions in the urban area served, e.g. pattern of indirect dischargers , product use pattern, user behaviour and pollutant load from urban surfaces (roofs, streets etc.) in case of combined sewer systems

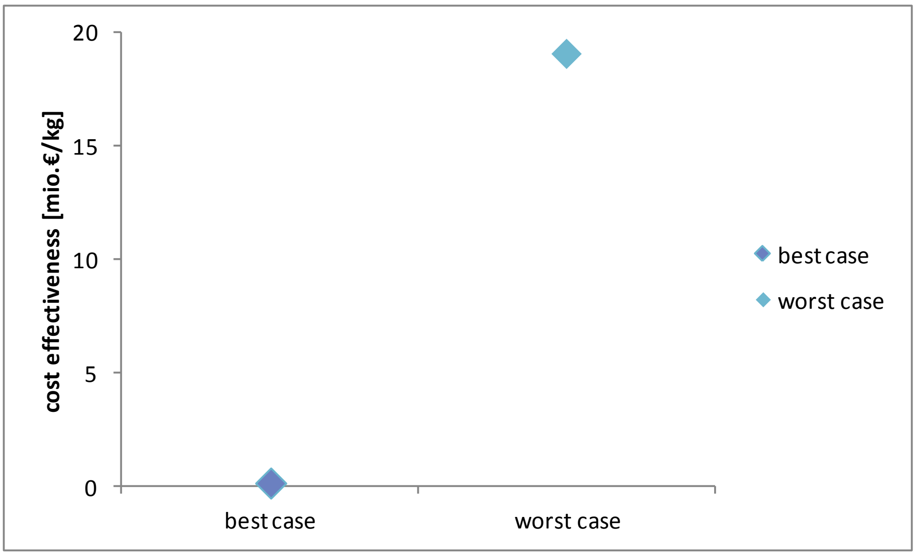
7.4.9 Political enforceability

The political enforceability depends on the willingness to pay in different countries. In Switzerland, use of PAC-adsorption at large MWWTPs has been mandatory since 2010.

7.4.10 Cost-effectiveness analysis

Taking into account typical loads, as discussed in 5.31, from 1.4 to 124 mg OP per year and capita, a reduction rate of 75 % using PAC and costs of 15 to 20 Euros per person for bigger MWWTPs, the cost-effectiveness lies between 160,000 and 19 million Euros per kg OP, see Figure 6.

Figure 6: Cost range for elimination of OP/OPE with AC treatment



7.5 Measure 5: Sludge treatment - controlled incineration

7.5.1 Description of source

As already shown above, OP has a high tendency to adsorb on sludge in MWWTPs. Fahlenkamp et al. (2006) calculated adsorption rates of 35, 55 and ca. 100 % of OP for 3 large MWWTPs in Germany. The corresponding concentrations in the stabilised sludge are between 4,600 to 80,000 µg/kg (median).

But of course the load of OP in sludge depends on its origin. If sludge is incinerated, use of the nutrients it contains , which is generally favoured, is not possible. Therefore, this measure cannot be recommended without looking at the local conditions and concentrations in sludge (ifeu, 2008).

7.5.2 Description of measure

To avoid contamination of soil by land-spreading of OP containing sludge, it is necessary to use incineration processes to degrade the substance. Different processes are currently in use, varying in reactor type (fixed or fluidized bed) or feed (mono- or co-incineration). The system mainly used for the incineration of sewage sludge is fluidized-bed combustion. Fluidized-bed furnaces for the incineration of sewage sludge are usually operated at combustion temperatures in the range of 850°C and 900°C.

7.5.3 Effectiveness

Because of the high temperature and the long residence time in the reactor OP is completely degraded. Commercially available installations are equipped with highly effective flue gas cleaning systems.

7.5.4 Costs

There are a number of sources in the literature that report different costs for co-incineration but the information on the costs of mono-incineration is more limited (Salado et al., 2008) . Sede and Andersen (2002) give the following estimates: for co-incineration: €290 t/dry sludge and for mono-incineration €374t/dry sludge. The costs of incineration are highly variable depending on design aspects and energy recovery, as sales of both electricity and heat can generate substantial revenue that can cover part of the incineration costs.

7.5.5 Secondary environmental effects

The main secondary benefit of sewage sludge incineration is the decomposition of all other organic substances like other HELCOM priority substances or micro pollutants like pharmaceuticals.

A big disadvantage lies in the fact that the phosphorus in the sewage sludge is lost for fertilisation purposes. There are R&D projects running on how to break the organic molecules and extract the nutrient in one process.

7.5.6 Technical feasibility

Specific sludge incineration facilities have been in operation in BSC countries for many years, i.e. in Germany and Denmark.

7.5.7 Secondary socio-economic effects (including indirect costs)

Incineration of sludge results in higher costs for MWWTPs and for the inhabitants they serve. Farmers can no longer use sludge as “cheap” fertiliser, in some cases they are paid for this “sludge disposal”.

7.5.8 Geographical and time scale of effects

According to a COHIBA survey, sludge incineration is used only in Denmark, Germany and Poland, see Table 9.

Table 8: Sludge going to incineration

	Year	Share of incineration
Denmark	2007	15.7%
Estonia	2009	0.0%
Finland	2008	0.0%
Germany	2009	52.6%
Latvia	2007	0.0%
Lithuania	2009	0.0%
Poland	2009	1.6%
Russia	2011	0.0%
Sweden	2009	0.0%

7.5.9 Political enforceability

The political enforceability depends on the willingness to pay in different countries. In Germany, some federal states decided politically to apply the precautionary principle and incinerate all their sludge.

7.5.10 Cost-effectiveness analysis

The costs for incineration of sludge are assumed to be 374 Euros per tonne of dry sludge. Based on concentrations between 4.6 and 80 mg/kg of sludge, the cost effectiveness equals 0.005 to 0.08 million Euros per kg of OP.

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VIII. Short-chain chlorinated paraffins (SCCP) and medium-chain chlorinated paraffins (MCCP)

8.1 Measure 1: Ban of SCCP in remaining areas of application. Ban/restriction of MCCP for usage in emulsion and oil based metal cutting fluids

8.1.1 Description of source

To avoid production, use and waste of products containing SCCP and MCCP, the use of MCCP in emulsion and oil based metal cutting fluids should be regulated, and regulatory measures on SCCP could be extended to processes where the use of SCCP has not been restricted so far.

8.1.2 Description of measure

The European Parliament has called for an extension of the ban on the use of SCCP in metal-working and leather processing applications to cover all applications, including use as a plasticizer in paints and coatings and flame retardants in rubber, plastics and textiles (Kesteven 2002), similar to the PARCOM phasing-out decision (1995) which is already being applied by several European countries (see Table 4).

Since MCCP have not been restricted so far, and since emissions from industry contribute to a large extent to the total load to the Baltic Sea area, the usage of MCCP in emulsion and oil based metal cutting fluids should be restricted, since this application is one of the main contributors to total emissions from industry.

8.1.3 Effectiveness

Regulations and even voluntary approaches that affected the main uses of SCCP have already been effective in the past. Sales in the EU decreased significantly over the years, as can be seen in Figure 7.

However, the effectiveness of regulations is limited if applied only to restricted regions (e.g. EU27). Products containing SCCP and/or MCCP can be produced outside these regions and imported (as long as import of SCCP/MCCP containing products itself is not restricted).

The production of SCCP is one good example of the limitations of regional restrictions: While the production of SCCP in EU27 has meanwhile decreased, it is reported that CPs of various chain lengths are produced in the USA, Russia, India, China, Japan and Brazil (ESWI 2011). For example, Jabr (2011) reports that China increased its production of SCCP by a factor of 30 within the last 20 years. And although Europe has restricted use of SCCPs, their manufacture is growing in China and possibly in India, raising concerns that worldwide exposure levels for people and wildlife might be increasing (Jabr 2011).

8.1.4 Costs

Costs generated will mainly be costs for substitutions for SCCP and MCCP and/or costs for product redesign to avoid use of SCCP and MCCP.

8.1.5 Secondary environmental effects

None, since this measure is substance specific.

8.1.6 Geographical and time scale of effects

As can be seen in Figure 7, regulatory and voluntary measures can lead to substantial reductions in occurrence of SCCP and MCCP over ten years. However, some of the SCCP and MCCP containing products might be in storage for several years. Additionally products such as conveyor belts have lifetimes of 12-22 years (ESWI 2011). Rubber products other than conveyor belts have an average lifetime of about 7.5 years. Therefore reductions of SCCP emissions into the environment can probably be measured with a delay of about 20-30 years.

8.1.7 Political enforceability

National or regional regulations can be effective for individual installations.

8.1.8 Cost-effectiveness analysis

Since the costs incurred will mainly be costs for substitutions for SCCP and MCCP and/or costs for product redesign to avoid the use of SCCP and MCCP the cost effectiveness of a ban on SCCP and MCCP depends on the costs for substitution. However, these cannot be stated in general.

Substitution of SCCP in rubber products and substitution of MCCP in emulsion and oil based metal cutting fluids.

8.1.9 Description of source

SCCP and MCCP are used in the rubber industry, in sealants and adhesives, paints and varnishes and in the textile industry due to their softening and flame retardant abilities. In metalworking MCCP are used as a replacement for SCCP due to their lubricating ability and as replacement for SCCP as fat liquors in leather processing.

8.1.10 Description of measure

To render products non-flammable in functions where this is required, flame retardants can be added to the material (as is the case with SCCP and MCCP in rubbers and textiles) or the product can be constructed in a way physically hindering flammability. Similar substitutions have to be found for the other areas of usage: plasticizer in paints and coatings, and in sealants and adhesives, and for lubrication in metal working and fat liquoring in leather processing.

Metal working

Since MCCP have similar characteristics to SCCP, they are used as replacements for SCCP e.g. as extreme pressure additives in metalworking fluids, as plasticizers in paints, and as additives in sealants (OSPAR 2001). In Sweden, long chain chlorinated paraffins (LCCPs) have been used in some demanding applications in metalworking fluids instead of SCCPs. LCCPs have also been suggested as a replacement for SCCPs in the leather industry as well as in paints and coatings, in sealants and rubber (OSPAR 2001).

Additionally other substitutes are available. For example, alkyl phosphate esters and sulfonated fatty acid esters may function as replacements for SCCPs as extreme pressure additives in metalworking fluids (Böhm 2003; OSPAR 2001).

Leather industry, paint and coatings, sealants

For the leather industry natural animal and vegetable oils can be used as alternatives to SCCP and MCCP containing fat liquors. In paints and coatings, phthalate esters, polyacrylic esters, diisobutyrate as well as phosphate and boron-containing compounds are suggested as replacements. Phthalates esters can be used as alternatives for use in sealants (OSPAR 2001).

Flame retardants

Alternatives as flame retardant in rubber, textiles and PVC are antimony trioxide, aluminum hydroxide, acrylic polymers and phosphate containing compounds. Sweden considers these substances as being less harmful than chlorinated paraffins. (OSPAR 2001)

Summary of alternatives

A summary of information on possible alternatives to SCCP is given in BRE et al. (2008) and listed in the Appendix in Table 9. However, there might still be uses for which these alternatives do not fulfill all technical and safety demands.

8.1.11 Effectiveness

Experience with avoiding SCCP and , partly, MCCP is available from several European countries that followed PARCOM decision 95/1 (see Böhm (2003) and Bolliger & Randegger-Vollrath (2003).

Substitutes such as e.g. sulfur based additives are available and often equally efficient as CP-based products (lubricants etc.). However, according to Böhm (2003) their suitability to serve as replacement is highly dependent on process conditions (temperature, friction, viscosity, process velocity).

8.1.12 Costs

The costs of some additives substituting SCCP and MCCP are in a comparable range (e.g. sulfur containing additives). However, additional costs can be incurred. Therefore final costs depend on the individual case and may vary. General information on costs cannot be given, neither now nor in the past (e.g. by Böhm 2003).

8.1.13 Secondary environmental effects

Due to the multitude of substitutes for the different uses in which SCCP and MCCP are to be replaced, a general statement regarding secondary environmental effects cannot be made.

Some of the substitutes are toxic as well or are ecologically problematic. Biodegradability is dependent upon the substitute's structure. According to an evaluation of different material safety data sheets (MSDSs) by Böhm (2003) biodegradation is possible for some products, however, for most products biodegradation is rather poor. General statements regarding toxic effects on aquatic organisms or bioaccumulation factors of substitutes cannot be made, based on MSDSs.

Additional information on the substitutes' behavior in the environment and on their degradability, generated in OECD standard tests, needs to be collected from the suppliers.

8.1.14 Technical feasibility

CPs can be substituted in nearly all of their applications. In quite a few European countries (including Germany and Switzerland) CPs have been replaced in most applications.

While CPs are generally applicable in a wide range of applications, for substitutes the specific boundary conditions and technical requirements have to be evaluated more thoroughly. From the large variety of potential substitutes the right components have to be chosen for each individual process. As Böhm (2003) stated: detailed experience is already available from some sectors of industry and can be transferred to other industrial areas (e.g. from automotive industry to other metal working industries).

A number of possible advantages (ecological and in terms of human health) which substitutes may have over CPs could not be evaluated by Böhm (2003) on the basis of available MSDSs.

8.1.15 Secondary socio-economic effects

Unknown

8.1.16 Geographical and time scale of effects

After development time, the measure becomes effective in reducing emissions of SCCP and MCCP (after replacement with substitute, emissions of substitute are possible). According to industry, innovation cycles may take up to 15 years. However, for most applications experiences are already available.

8.1.17 Political enforceability

Substitution of SCCP and MCCP is possible in most applications and has already been completed in some European countries due to political enforcement.

8.1.18 Cost-effectiveness analysis

Due to the wide range of applications where SCCP and MCCP have to be substituted, and due to the even wider variety of substitutes, a general statement cannot be made.

8.2 Measure 2: Advanced waste water treatment – AC treatment

8.2.1 Description of source

The kinds and loads of pollutants in waste water vary greatly between different cities/districts/MWWTPs, and can also vary markedly in time. Therefore, predicting what kinds and loads of pollutants are treated at MWWTPs has a very high uncertainty.

8.2.2 Description of measure

A prerequisite for advanced wastewater treatment and also for sludge treatment (see chapter 6.6) is mechanical and biological wastewater treatment because in this treatment SCCP and MCCP accumulates in sludge which then has to be further treated. It is expected that about 90-93% of SCCP and even more of MCCP ends up in sewage sludge. The remaining 7-10% are expected to pass through the wastewater treatment plant and have to be treated by advanced wastewater treatment steps, such as activated carbon (see chapter 6.3), membrane filtration (see chapter 6.4), and oxidative techniques (see chapter 6.5), in cases where relevant loads are emitted. Membrane technology, but also filtration methods using activated carbon are able to improve the elimination rates of municipal wastewater treatment plants with regard to various priority substances (Hillenbrand et al. 2007).

Activated Carbon (AC) filters (AC) are a proven technology for removal of pollutants from wastewater. AC has a large surface area and is an effective sorbant for many substances. Different technical systems are commercially available (e.g. powder (PAC) and granular activated carbon (GAC)).

8.2.3 Effectiveness

Advanced wastewater treatment steps will most likely have a positive effect on the reduction of SCCP and MCCP emissions. However, data are scarce.

The effectiveness of AC filters at MWWTPs for elimination of pollutants depends on the concentration range of the pollutant, technical parameters and the matrix. Spent activated carbon with adsorbed SCCP and MCCP has to be incinerated (LANUV 2008).

The measure has major cross substance effects, e.g. removal of TBT, PFOS, nonylphenol, Cd, Hg and other pollutants.

8.2.4 Costs

Economic analysis in the “Strategie MicroPoll” project (Switzerland) found costs of EUR 10-60 per person and year, including discounted investment costs and running costs (Sterkele & Gujer 2009). Specific costs are strongly dependent on the size of the MWWTP (large WWTPs show low specific costs, economies of scale).

8.2.5 Secondary environmental effects

AC filtration at MWWTPs, which is sometimes called the 4th stage of waste water treatment, affects emissions of all HS of special concern to the Baltic Sea, which are typically present in municipal waste water in low concentrations. It also has potentially large positive water related secondary environmental effects: on phosphate emissions and on other pollutants, such as heavy metals, organic micropollutants (which are not on the HELCOM list), pharmaceuticals and their metabolites, or endocrine disrupters. Negative secondary environmental effects are related to e.g. energy use and GHG emissions during construction and operation, and for manufacture of activated carbon (Wenzel et al. 2008).

8.2.6 Technical feasibility

- Proven technology
- Effectiveness depends on matrix (e.g. COD, other micropollutants), concentration of sorbent, bed exchange rate
- Filter material contains HS and must be handled accordingly (e.g. incinerated, not included in example calculation)

Limitations: The technical prerequisite is a well-functioning MWWT with low concentrations of suspended solids and dissolved organics (BOD and COD), skilled personnel required for O&M (but not different from O&M of large MWWTPs), handling of wastes.

8.2.7 Secondary socio-economic effects (including indirect costs)

If large MWWTPs are equipped with a “4th stage”, the question is who pays for it. One option is that the respective MWWTPs charge the costs to their clients. Large MWWTPs often have lower per capita costs than smaller MWWTPs.

The other option is to have the costs paid for by all citizens (via taxes), as the whole community benefits from a toxfree environment. The latter option was put into practice in Switzerland (total costs of waste water treatment rose by 6%). A very good option, but hard to put into practice, is to tax production and use of products containing hazardous substances. This would follow the polluter pays principle.

8.2.8 Geographical and time scale of effects

This measure becomes effective immediately and has a technical life span of 80 years for built infrastructure. It seems sensible to target large plants because of economies of scale effects, which are usually located near large settlements. In the context of the projects, MWWTPs near the shore can be interesting targets.

8.2.9 Political enforceability

In Switzerland AC treatment has been mandatory for large MWWTPs since 2010.

8.2.10 Cost-effectiveness analysis

From detected concentrations of SCCP and MCCP in the effluent and sludge of wastewater treatment plants two scenarios are derived for each:

- low SCCP load in regular effluent: 16 mg/cap*a
- high SCCP load in regular effluent: 113 mg/cap*a
- low SCCP load in regular sludge: 13 mg/cap*a
- high SCCP load in regular sludge: 42 mg/cap*a
- low MCCP load in regular effluent: 79 mg/cap*a
- high MCCP load in regular effluent: 612 mg/cap*a
- low MCCP load in regular sludge: no data available
- high MCCP load in regular sludge: no data available

The effectiveness of AC filters in removing SCCP and MCCP from municipal waste water is assumed to be high with 90-99%. Results of Nielsen et al. (2011) support this assumption.

Costs are assumed to be between 15 to 20 €/cap*a for a MWWTP with 500 000 p.e., based on a cost study from Switzerland .

The costs for removing SCCP by activated carbon filtration as a fourth step in a MWWTP would vary within a range of 0.13-1.67 Mio. €/kg removed product, and between 0.02-0.34 Mio. €/kg removed product for MCCP.

8.3 Measure 3: Advanced waste water treatment - membrane filtration

8.3.1 Description of source

See chapter 6.3.1.

8.3.2 Description of measure

Membrane technology is a physical separation process which filters out particles of varying size depending on the size of the pores of the membrane. Corresponding to the separation dimensions, a distinction is made between micro-, ultra- and nanofiltration and reverse osmosis. In industry, membrane technology is already used on a large scale to separate substances as well as to treat wastewater. It has only been used in municipal sewage treatment for a few years; the first large-scale plant was put into operation in Germany in 1999 (WWTP Rödingen) (Hillenbrand et al. 2007).

Membrane technology can be applied at two points: integrated into the activated stage to substitute conventional final clarification for separating the activated sludge (membrane activated sludge process), or downstream from conventional final clarification for advanced treatment of the effluent.

Micro- or ultra-filtration membranes are often used in membrane separation activated sludge processes. According to information from the DWA-Fachausschuss KA-7 (2005), the maximum separation limit is 0.4 µm, but modules are sometimes used with a pore size as small as 0.04 µm (Vossenkaul 2005).

The most important reason for using membranes downstream from a conventional sewage plant is usually the obvious improvement in effluent quality (Lange et al. 2006). Pollutants, micro organisms and even viruses can be retained to the extent that they are attached to larger particles.

8.3.3 Effectiveness

Reliable data regarding effectiveness of membrane filtration for removal of SCCP and MCCP are not available but it is assumed that retention rates are in a range of 75-99%.

8.3.4 Costs

The membranes themselves account for a large share of the total investment. However, there has been a clear drop in costs in the last few years due to learning and economies of scale effects (Hillenbrand & Hiessl 2006), and costs can be expected to drop further in the future. Technical simplifications are also expected regarding the mechanical equipment and its incorporation into the total system (DWA-Fachausschuss KA-7 (2005) cited in Hillenbrand et al. (2007)).

The costs of the membrane play a significant role since the service life of the membrane is generally much shorter than the depreciable life of the machine technology. Attempts are being made to reduce the costs of replacing membranes by both lowering the specific membrane costs and lengthening the service period (target: 7 to 10 years). Energy costs (specific total energy consumption approx. 0.8 to 1.6 kWh/m³, sometimes even up to 2.0 kWh/m³, compared to 0.3 to 0.5 kWh/m³ in conventional systems without water disinfection; (DWA-Fachausschuss KA-7 (2005) and Krampe & Laufer (2005), both cited in Hillenbrand et al. (2007); (Lange et al. 2006; Wedi 2005)), and the costs for needed chemicals also make up a significant part of the operating costs. Table 6 gives an overview of the various cost shares of membrane activated sludge systems which illustrates that the cost for replacing the membrane is of overriding significance. A comparison of operating costs based on quotes showed operating costs of 0.24 to 0.25 €/m³ for the largest currently operated municipal membrane activated sludge plant (KA Nordkanal) which is approx. 15 % higher than the conventional solution (0.20 to 0.22 €/m³, Engelhardt (2002)).

Table 6: Overview of the various cost shares of activated membrane plants (DWA-Fachausschuss KA-7 (2005) cited in Hillenbrand et al. (2007))

		Costs [Ct/m^3]	Category[*1)]
Crossflow aeration	0.20 – 0.75 kWh/m^3	2.0 – 7.5	O
Permeate/recirculation	0,08 – 0.10 kWh/m^3	0.8 – 1.0	O
Additional aeration demand	0.08 – 0.10 kWh/m^3	0.8 – 1.0	O
Chemicals	0.20 – 1.10 €/m^2 a	0.3 – 1.8	O
Membrane replacement	10 – 5 a	13.3 – 26.6	C

*1): O = operating costs; C = capacity cost

Electricity: 10 Ct/kWh; resultant sewage 90 m3/(resident ● a), spec. membrane area: 1.5 m2/resident, usual market costs for H2O2, acids and bases, membrane costs 80 €/m2

The costs of introducing a downstream membrane stage are between 0.25 €/m³ and 0.42 €/m³ filtrate, according to results of pilot studies (Dittrich et al. 1998). First estimates put the costs of activated carbon treatment including grit filtration at about 10 cents per m³ or at 6 cents if a grit filter is already fitted (Neifer & Krampe 2006).

8.3.5 Secondary environmental effects

Additionally, membrane separation activated sludge treatment has operational advantages compared with conventional sewage plants since higher concentrations of dry solids can be suspended in the aeration tanks and thus higher concentrations of micro organisms. As a result, not only does the conventional final sedimentation stage become superfluous, but the activated stage can also be scaled down.

8.3.6 Technical feasibility

In Germany, experience has only been gained with a few large-scale systems. According to (Lange et al. 2006), 3 plants are currently in operation – partly in research and development projects (Geiselbullach, Merklingen, and Bondorf-Hailfingen).

8.3.7 Secondary socio-economic effects

For general effects see chapter 6.3.7.

The particular advantages of the membrane method of wastewater treatment are

- complete retention of solids and, as a result, an improved effluent quality with respect to the parameters COD and BSB5; hygienic effluent (i. e. filtration and decontamination system in one stage); effluent quality not affected by floating sludge, bulking sludge or foam formation (improvement of operational reliability),
- the demands made on advanced wastewater treatment concerning the protection of water and groundwater can be fulfilled because of the high purification capacity (e. g. lower pollutant concentrations, avoidance of floating sludge),
- the plants are easy to expand because of their modular , and they can be used in both large, municipal wastewater treatment plants and small, decentralized ones.
- Drawbacks include
- the higher operating costs caused by higher energy costs and higher costs for maintenance of the membrane modules,
- the more complex preliminary mechanical treatment necessary to protect the membrane,
- the additional demands made on process control as well as

- the greater sensitivity of the membrane to shock loads.

  (DWA-Fachausschuss KA-7 2005; MUNLV 2003)

Particulates are largely removed in accordance with the functional principle of membrane technology. This also removes any pollutants adhering to these particles. For example, heavy metals or PAHs show a high adsorption tendency. Böhm et al. (2002) estimated that membrane filtration can remove an additional 30 to 70% of heavy metals present in waste water. However, so far, there are little detailed studies available on the additional elimination capacity of membrane technology in municipal sewage plants. But in principle, activated sludge plants with membrane filtration can also improve the elimination of organic, non-readily degradable pollutants. This is achieved because the biocenosis which is formed in a plant with a high sludge age is better adapted to pollutants present in low concentrations. This aspect is being examined especially in connection with the emissions of endocrine substances from municipal sewage plants (Hegemann et al. 2002; Schiewer et al. 2001; Schröder 2003).

Besides the retention of priority pollutants, other additional water-relevant effects should be noted which may be of relevance within the scope of a comprehensive river basin management:

- wastewater disinfection (especially relevant if the water continues to be used, e. g. for recreational purposes),
- almost complete removal of particulate substances and the phosphorus bound to them (the reduction of particulates in wastewater also decreases the formation of sludge and sediment in the water, i e. improvement of natural habitats),
- improved degradability of organic trace elements when using membrane biology, corresponding at least to the share adsorbed by particulates, and
- possible further use of the purified wastewater as service or process water.

Additionally, membrane separation technology has the advantage over water disinfection using ultraviolet treatment, ozonization or chlorination that no unwanted by-products are formed.

8.3.8 Geographical and time scale of effects

See chapter 8.2.8.

8.3.9 Political enforceability

To enforce the use of membrane technology in wastewater treatment specific water protection requirements can be taken into account when issuing plant permits (Hillenbrand et al. 2007). Subsequent utilization requirements (bathing water, other recreational activities) also should be considered.

8.3.10 Cost-effectiveness analysis

Compared with conventional municipal sewage plants, activated sludge treatment with membranes greatly improves effluent outlet qualities. The additional investments required for this consist of the actual membrane itself, more powerful aeration, chemical and dosing systems as well as more complex mechanical preliminary treatment. These costs are set against the savings that arise when the membrane treatment eliminates the need for the final clarification stage and part of the aeration tank volume – because higher solid concentrations are possible in the activated stage. In addition, membrane separation activated sludge treatment may be advantageous in terms of the simplified possibilities for sludge stabilization as well as the reduced space required. As a consequence additional investments for implementation of this step might be balanced, depending on the given local conditions (Lange et al. 2006; Wedi 2005).

Based on the estimated loads in chapter 8.2.10, an assumed effectiveness of membrane filtration of 75-95%, and estimated costs between 25 to 42 €/cap*a, the costs for removing SCCP by membrane filtration as a fourth step in a MWWTP would vary in a range of 0.23-3.50 Mio. €/kg removed product, and between 0.04-0.71 Mio. €/kg removed product for MCCP.

8.4 Measure 4: Advanced waste water treatment - oxidative techniques

8.4.1 Description of source

See chapter 8.2.1.

8.4.2 Description of measure

Different oxidative techniques are available, e.g. ozonation, radiation by UV light. Long chain molecules are attacked and oxidized either directly by very reactive ozone molecules or indirectly by radicals generated by the process.

8.4.3 Effectiveness

Reliable data regarding the effectiveness of oxidative techniques for removal of SCCP and MCCP are scarce.

8.4.4 Costs

Reliable data regarding the costs of oxidative techniques for removal of SCCP and MCCP are not available.

8.4.5 Secondary environmental effects

See chapter 8.3.5.

Due to the presence of chlorine in SCCP and MCCP AOX could form. However, reliable data are not available.

8.4.6 Technical feasibility

Reliable data regarding the technical feasibility of oxidative techniques for removal of SCCP and MCCP are not available.

8.4.7 Secondary socio-economic effects

See chapter 6.3.7.

8.4.8 Geographical and time scale of effects

See chapter 6.3.8.

8.4.9 Political enforceability

To enforce the use of oxidative techniques in wastewater treatment specific water protection requirements can be taken into account when issuing plant permits (Hillenbrand et al. 2007). Subsequent utilization requirements (bathing water, other recreational activities) also have to be considered.

8.4.10 Cost-effectiveness analysis

Since reliable data are not available, neither on costs nor on effectiveness, a cost-effectiveness analysis cannot be conducted at present.

8.5 Measure 5: Sludge treatment - Controlled incineration

8.5.1 Description of source

It is expected that about 90% to 93% of SCCP and MCCP is adsorbed on sewage sludge and the rest stays in the water phase (Bolliger & Randegger-Vollrath 2003).

8.5.2 Description of measure

About 50% of sewage sludge from waste water treatment plants in EU27 is recycled, which means application to land in most of the cases. About 16%each is landfilled, incinerated, or treated otherwise (e.g. exported). Of these measures only controlled incineration of sludge is effective for the control of hazardous substances contained in sludge. Incineration of sewage sludge can be conducted as co-incineration in power plants or in the cement industry, or in sludge incineration plants, so called “mono-incineration”. The technical process is similar to the incineration of waste with temperatures above 850°C and flue gas treatment.

8.5.3 Effectiveness

Due to decomposition of SCCP and MCCP at high temperatures (above 200°C for SCCP), incineration of SCCP and MCCP containing sludge will most likely be effective. Although reliable data are not available, it is assumed that removal rates of 90-100% can be reached.

8.5.4 Costs

Costs for sewage sludge mono-incineration vary depending on size, location and other conditions. For a mono-incineration plant with a capacity of 30,000 Mg dm/a Schaum et al. (2010) reported investment costs of EUR 64-69 million (EUR 43-46 per Mg dry matter over a lifetime of 50 years without interests etc.), and operating costs of EUR 235 per Mg dry matter. For co-incineration investment costs would be in a range of EUR 34-43 million (not including costs for incineration plant).

8.5.5 Secondary environmental effects

Since the process has similarities to the incineration of waste, see chapter 8.6.

Additionally, due to the possibility of phosphorus recovery from ashes, mono-incineration plants should be favored.

8.5.6 Technical feasibility

Process is already well proven on a large scale.

8.5.7 Secondary socio-economic effects

Since this process is a main measure for all HS present in sludge, see general COHIBA recommendation report for details.

8.5.8 Geographical and time scale of effects

Since this process is a main measure for all HS present in sludge, see general COHIBA recommendation report for details.

8.5.9 Political enforceability

Since this process is a main measure for all HS present in sludge, see general COHIBA recommendation report for details.

8.5.10 Cost-effectiveness analysis

Based on average loads of SCCP in sludge between 13-42 mg per person and year, an assumed effectiveness of sludge incineration of 100%, and estimated costs between EUR 8 to 9.8/cap*a, the costs for removing SCCP by sludge incineration would vary in a range of 0.19-0.75 Mio. €/kg removed product. For MCCP reliable data about loads in sludge are not available.

8.6 Measure 6: Waste management – Controlled landfilling / controlled incineration

8.6.1 Description of source

Disposal of SCCP and MCCP containing products to controlled landfill sites is possible. Preferred waste management option for waste conveyor belts, gaskets and hoses, sealants and adhesives, and textiles.

ESWI (2011) gives a detailed overview of SCCP mass flows from relevant sources to current disposal/recovery operations in the EU27. According to this reference, the main source of SCCP in products are currently conveyor belts for underground mining, since SCCP are not restricted in this use due to their special characteristics (softener and flame retardant) that are needed in these products. Generally, emissions from rubber products account for about 60%, waste from sealants and adhesives for about 21% and paints and varnishes for about 15% of all SCCP containing waste in EU27.

8.6.2 Description of measure

Controlled landfill means that the site needs to be fitted with a leachate barrier system and leachate collection system.

In controlled incineration waste is burned at temperatures of 850-950˚C. Flue gases that are generated during the process are treated to reduce the amounts of hazardous substances before being emitted (ECB European Chemicals Bureau, 2008). Since SCCP decomposes at temperatures above 200 °C (Bolliger & Randegger-Vollrath 2003), the majority of SCCP in the waste is therefore decomposed during incineration. Emissions of SCCP from incineration plants are believed to be negligible.

8.6.3 Effectiveness

It is assumed that SCCP are largely destroyed during incineration at conditions in accordance with Directive 2000/76/EC. However, no exact figures on removal potential have been found in the literature. It is therefore assumed that 90-100% of the SCCP is decomposed during incineration.

8.6.4 Costs

Landfill has always been the cheapest waste disposal option, but costs of landfill as waste disposal option are rising rapidly. The situation as well as the cost incurred by handling of hazardous waste varies from landfill to landfill and according to boundary conditions.

Landfill costs vary from EUR 150-200/m3 (Svedberg Bo et al., 2010), EUR 260-350/t (EC, 2002) to EUR 309/t (RPA, 2008).

8.6.5 Secondary environmental effects

Chlorinated paraffins can act as a source of chlorine radicals during incineration processes which can lead to the formation of polychlorinated dioxins and furans. Since this formation is a well known problem with incineration, controls are already in place in most cases (EC 2008). Additionally SCCP could be a possible source of PCB (polychlorinated biphenyls) and PCN (polychlorinated naphthalenes) formation in waste incineration (OSPAR 2009).

8.6.6 Technical feasibility

Controlled landfilling and controlled incineration of waste are well proven and often applied technologies. Thus, technical feasibility is not the limitation but the collection of waste products can be a large challenge.

8.6.7 Secondary socio-economic effects (including indirect costs)

Lack of landfill space increases the costs. After closure all leak detection systems and groundwater monitoring systems are required to be maintained. Post-closure care usually lasts for 30 years after closure.

8.6.8 Geographical and time scale of effects

Landfilling is not preferred in all countries because of a lack of land.

8.6.9 Political enforceability

Since this process is a main measure for all HS present in sludge, see general COHIBA recommendation report for details.

8.6.10 Cost-effectiveness analysis

SCCP and MCCP loads in waste cannot be clearly determined. The amount of SCCP and MCCP in waste depends largely on which waste fractions are collected and landfilled. If waste conveyor belts, gaskets and hoses, sealants and adhesives, and textiles are collected separately, the load of SCCP and MCCP is higher than would be the case if a mixture of household waste and the named rubber wastes is collected.

The load of SCCP in waste and the absolute amount of SCCP that is decomposed as a result of incineration of one tonne of waste cannot be clearly determined. The amount of SCCP in incinerated waste depends largely on which waste fractions are burned. If only waste conveyor belts, gaskets and hoses, sealants and adhesives, and textiles are burned, the load of SCCP would be higher than if a mixture of household and the named rubber wastes is burned.

Due to these unknown facts, cost effectiveness cannot be evaluated at present.

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IX. Endosulfan

9.1 Measure 1: Ban and substitution of endosulfan

9.1.1 Description of source

Current annual production and use amount to between 18,000 to 20,000 tonnes worldwide outside the EU and Baltic Sea Region. About 75% of endosulfan load reaches the Baltic Sea Region as atmospheric deposition, produced and used mostly in the other regions. Another source can be imported foodstuff from countries using this insecticide.

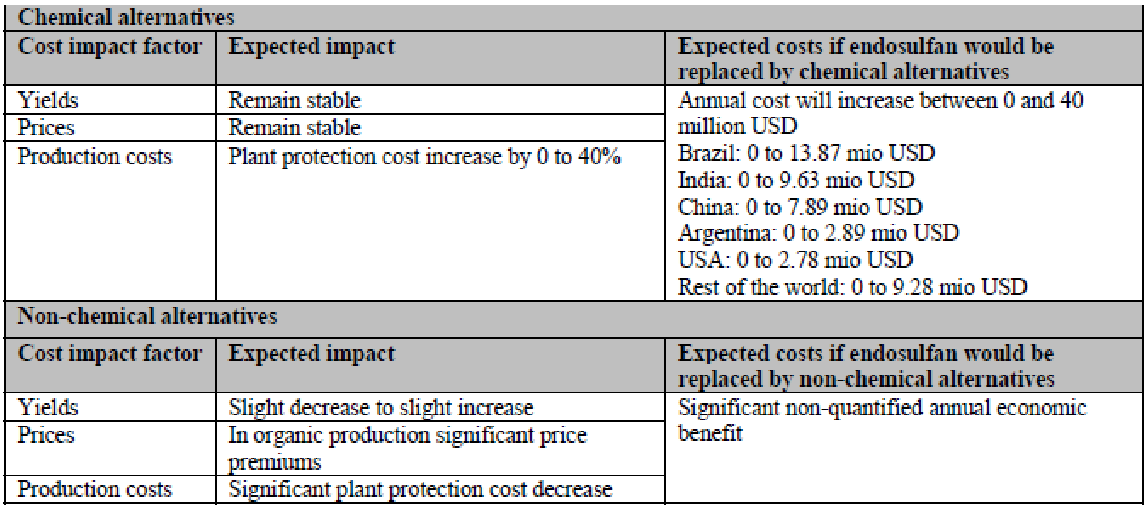
9.1.2 Description of measures

Because of endosulfan threats to the environment, also due to the fact that substitutes for endosulfan are available in sufficient quantities, a global ban on the use and manufacture of endosulfan was agreed under the Stockholm Convention with some exempts[10]. The use of endosulfan can be replaced by several chemical and non-chemical alternatives. A wide range of crop-pest complexes exist and for each specific crop-pest complex an appropriate combination of chemical, biological and agricultural control action may be taken. Statements that alternatives do not exist for specific crop-pest complexes may be based on considerations that are focused only on chemical alternatives and may not always appropriately consider non-chemical control measures. In specific cases, promising research on semio-chemicals is ongoing and may be used in the foreseeable future.

9.1.3 Costs

Negative annual cost impact due to increased plant protection costs ranges between 0 and 40 million USD (for Brazil: 0 to 13.87 million USD, for India: 0 to 9.63 million USD, for China: 0 to 7.89 million USD, for Argentina: 0 to 2.89 million USD, for the USA: 0 to2.78. million USD and for the rest of the world: 0 to 9.28 million USD), if endosulfan will be replaced by chemical alternatives. In contrast non-quantified positive annual cost impacts are possible if endosulfan will be replaced by non-chemical alternatives.

Table 4: Expected economic impacts on agriculture if endosulfan will be replaced by chemical and non-chemical alternatives



Chemical alternatives		
Cost impact factor	**Expected impact**	**Expected costs if endosulfan would be replaced by chemical alternatives**
Yields	Remain stable	Annual cost will increase between 0 and 40 million USD Brazil: 0 to 13.87 mio USD India: 0 to 9.63 mio USD China: 0 to 7.89 mio USD Argentina: 0 to 2.89 mio USD USA: 0 to 2.78 mio USD Rest of the world: 0 to 9.28 mio USD
Prices	Remain stable	
Production costs	Plant protection cost increase by 0 to 40%	
Non-chemical alternatives		
Cost impact factor	**Expected impact**	**Expected costs if endosulfan would be replaced by non-chemical alternatives**
Yields	Slight decrease to slight increase	Significant non-quantified annual economic benefit
Prices	In organic production significant price premiums	
Production costs	Significant plant protection cost decrease	

9.1.4 Technical feasibility

Technically feasible substitutes for endosulfan have been identified in both developed and developing countries.

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[10] At its fifth meeting held from 25 to 29 May 2011, the Conference of the Parties adopted an amendment to Annex A to the Stockholm Convention to list technical endosulfan and its related isomers with a specific exemption (decision SC-5/3). Pursuant to paragraph 4 of Article 21 of the Convention, the amendment was communicated by the depositary to all Parties on 27 October 2011

9.1.5 Cost-effectiveness analysis

Due to lack of data in terms of the global ban and substitution of endosulfan, it is not possible to perform a cost-effectiveness analysis.

9.1.6 Geographical and time scale of effects

The effect of this measure is important at the international level.

9.1.7 Political enforceability

Conflicting interests between the EU and countries still producing and using endosulfan can be expected.

9.2 Measure 2: Sludge treatment – Controlled incineration

9.2.1 Description of source

Endosulfan is partly emitted through diffuse sources, such as imported foodstuffs from countries still using endosulfan as a plant pesticide and emissions during the manufacture of food products and from household consumption of foodstuffs. Due to the physical and chemical properties of endosulfan, the substance is adsorbed to the solid fraction in the aquatic environment. Thus, it is crucial to correctly manage the matrix where most of these substances accumulate – sludge of WWTPs.

9.2.2 Description of measure

The options to deal with sludge vary from depositing them in landfills, to applying them to agricultural and forest soils as a fertilizer, even up to incineration in hazardous waste incinerators. Incineration is usually done in specially designated facilities, which can be either dedicated only to sludge incineration in cases when incoming loads are quantitatively large, or can incinerate other types of hazardous waste as well. The most common technologies used for sludge incineration are hearth incineration and fluidised bed incineration. Each system has its advantages – hearth incinerators are the most widely used at the moment – the movement of sludge through hearths exposes a larger area for incineration than other technologies, making it very efficient. Fluidised bed incinerators, however, allow high utilisation of energy, thus leading to cost saving. In general, the concrete technology should be chosen depending on site-specific and local conditions.

9.2.3 Effectiveness

From the perspective of reducing potential emissions, incineration is the most effective option, as it practically eliminates all traces of most organic pollutants in the sludge thus preventing any further emissions. Regular monitoring of emission gases from the incineration process must be conducted, but usually such installations are equipped with advanced air purification devices that allow all air quality requirements to be met. Residue of endosulfan stocks can be destroyed by incineration at 900°C with a flux of 300 kg per hour

9.2.4 Costs

Incineration can be a very costly option, especially if no local/regional incinerators are available. Costs can vary considerably, depending on specific local and regional cases. Generally speaking, the highest costs are the investments to build an incineration facility. Operating costs can vary depending on amounts of sludge incinerated, moisture of sludge, whether there are other types of waste incinerated as well, etc. In the long term, investments may pay back and become more feasible, especially if the incinerator can generate heat or electricity which can be sold to consumers, and if sludge would otherwise have to be transported over long distances or exported for utilisation.

9.2.5 Geographical and time scale of effects

Endosulfan was used mostly in the western part of the Baltic Sea Region, therefore problems with residues in households and controlled landfilling of sludge arise mostly in the western part.

9.3 Measure 3: Advanced waste water treatment – AC treatment

9.3.1 Description of source

As endosulfan emissions can come from diffuse sources, emissions during the manufacture of food products and from household consumption of foodstuffs, endosulfan can also be found in wastewater in food processing industries. Endosulfan is adsorbed to the solid fraction in the aquatic environment, but can also be found in the water phase.

9.3.2 Description of measure

Activated carbon filters (AC) to remove pollutants from wastewater are a proven technology. AC has a high surface area and is an effective sorbant for many substances. Different technical systems are commercially available (e.g. powder (PAC) and granular activated carbon (GAC)).

9.3.3 Effectiveness

The use of activated carbon filters can effectively remove 75-98% of endosulfan from waste water. The measure has large cross-substance effects, eliminating other of the 11 hazardous substances.

9.3.4 Costs

Economic analysis from the project “StrategyMicroPoll” (Switzerland) found costs of € 10-60 per person and year, including discounted investment costs and running costs. Specific costs strongly depend on the size of MWWTP (large WWT, low specific costs, economies of scale).

9.3.5 Secondary environmental effects

AC filter in MWWTP, sometimes called the 4th stage of waste water treatment, affects endosulfan emissions as well as emission of the other 11 HS of special concern to the Baltic Sea, which are typically present in municipal wastewater in very low concentrations. It also has potentially large positive water-related secondary environmental effects: on phosphate emissions and on other pollutants, such as heavy metals, organic micropollutants (which are not on the HELCOM list), pharmaceuticals and their metabolites, or endocrine disrupters. Negative secondary environmental effects are related to e.g. energy use and GHG emissions during construction and operation, and for manufacturing activated carbon.

9.3.6 Technical feasibility

- Proven technology
- Effectiveness depends on matrix (e.g. COD, other micropollutants), concentration of sorbant, bed exchange rate
- Filter material contains HS and must be handled accordingly (e.g. incinerated, not included in example calculation)

The technical prerequisites for AC treatment is a well-functioning MWWT with low concentrations of suspended solids and dissolved organics (BOD and COD), skilled personnel are required for O&M (but not different from O&M of large MWWTP) and handling of wastes.

9.4 Measure 4: Improvement of control and ban of contaminated foodstuff

9.4.1 Description of source

Endosulfan is still used in several countries, mostly developing countries in tropical regions, due to the lower price of endosulfan compared to the other pesticides and alternatives. Contaminated foodstuffs do not only originate from developing countries, but can be found in food also from countries within the EU – despite the current ban of the substance.

9.4.2 Description of measure

It is important to avoid endosulfan in our environment by controlling and monitoring imported foodstuffs from countries where endosulfan is still used. Developing control and monitoring capacity for laboratories of the food agency/health board to enable analysis of endosulfan at the best level is important.

9.4.3 Effectiveness

Effectiveness of the measure is 100%, as this measure is precautionary and avoids the occurrence of endosulfan in Baltic Sea Region.

9.4.4 Costs

Main costs of the measure depend on laboratory costs for technology and staff and for organising sampling. Costs depend on the cost-effectiveness of laboratories, number of samples, etc.

9.4.5 Secondary environmental effects

Secondary environmental effects can be a reduction of other pesticides in foodstuffs.

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X. Mercury (Hg)

10.1 Measure 1: Substitution of mercury in products and processes

10.1.1 Description of source

Mercury is used in various applications. Although product-oriented policies have been implemented to phase out individuals' use of mercury-based products, e.g. thermometers; in industry and in medicine, mercury can also be emitted from the following products and processes:

1. mercury, mercury compounds and mixtures containing mercury
2. products containing mercury and
3. industrial processes involving mercury

10.1.2 Description of measure

Substitution as a source control option limits the emissions of mercury by producers in various production processes, including limiting surface runoff, controlling point source pollution and implementing "good housekeeping" practices. Mercury has been successfully substituted in many applications and products, especially in batteries, thermometers, wiring and electrical equipment. The main possible substitutions are in the field of dentistry, which are very important due to their life cycle impacts.

10.1.3 Effectiveness

Mercury substitution is effective in eliminating the risk to water by lowering the impact on the whole cycle of the substance. Effective substitution is as follows: dental amalgam 50%, batteries 50-100%, light sources ca. 25%, measuring equipment 75 – 100%, chemicals 50% - 100%, chlor-alkali up to 100%. In terms of dental equipment and supplies, the potential for reducing mercury input is 80-110 t Hg per year.

10.1.4 Costs

In terms of dental equipment and supplies, the substitute has its price, whereby these costs are transferred to the patients. Replacing the amalgam filing by dentists is estimated at 129 000 US$/kg of Hg (UNEP, 2010). The cost to the end-user of reduced mercury input amounts to 11,000 - 78,000 €/ kg Hg.

The cost of abatement techniques for crematoria in the Netherlands ranges from € 27,270 to € 48,180 per year for cold start furnaces and from € 45,460 to € 74,550 for hot start furnaces. Costs will most likely be passed on to the public.

Costs for containing thermometers (0.05-5 g) are estimated to be $10-710, liquid in glass thermometers < $15, digital thermometers $12. Industrial thermometers: $10-60, liquid-filled $ 2-138, digital $ 14-260. The alternative for Hg in manometers ($20-375) is Needle/bourdon with costs ranging from $50 to $250.

10.1.5 Technical feasibility

The technical feasibility of the substitution and good practice options are presented in Table 6. Improvements in process technologies and end-of-pipe measures are generally feasible but facility-specific conditions should be taken into account, e.g. additional infrastructure might be required.

Table 6: Technical feasibility of mercury substitution and good practices of product use

Type of use	Substitution	Good practices
Dental equipment and supplies	Feasible, various possible substitutes. Replacement of the amalgam fillings at dentists and their removal after death can be restricted due to cultural	-BMP greatly reduces mercury emissions to water from dentistry - Pre-treatment techniques for dental wastewater - Abatement techniques for

	reasons	crematoria in use at a number of installations in Europe
Measuring and control equipment	Exists e.g. thermometers	Used in industry so there are good opportunities to manage the wastes
Batteries	Feasible with exemptions, possible limitation of Hg content	Mercury-based batteries are used in industry so there are good opportunities to manage the wastes
Wiring devices and switching equipment	Feasible with exemptions	Used in industry so there are good opportunities to manage the wastes
Pharmaceuticals	Already applied	Pre-treatment methods are used in a few cases
Electric lighting, luminescent lights	Feasible with exemptions	Used in industrial and commercial applications. High uncertainty concerning the waste management
Laboratories	Feasible with exemptions	Good opportunities to manage the wastes

10.1.6 Geographical and time scale of effects

Substitution of mercury in products is relevant for the entire Baltic Sea Region; and mostly in western parts of the BSR for dental amalgam.

10.1.7 Political enforceability

Despite existing regulations and policy control, institutional problems concerning efficient implementation, monitoring and control can be expected on a country or regional level.

10.1.8 Cost-effectiveness analysis

The cost-effectiveness of amalgam separators ranges from US$33,000 to 1,300,000 /kg Hg removed. Replacing or removing the filling at death is estimated at US$400 /kg of Hg (Sweden). Flue gas cleaning with carbon in crematoria is estimated at US$29,000 – 340,000 /kg Hg (Sweden, UK) (Hylander and Goodsite, 2006).

10.2 Measure 2: Improvement of BAT and revision of BREF for combustion power plants

10.2.1 Description of source

Coal combustion in power plants and industrial boilers, cement production, non-ferrous metal production (incl. gold) and waste incineration together contributed about 51 per cent to the total global emissions of mercury to air in 2005. The magnitude of the mercury emissions from these sectors depends on a number of factors including the mercury content and other compositional characteristics of fuels and raw materials, the technical configuration of the source facility and the presence of emission control equipment for air pollutants and mercury-specific controls. Mercury is emitted from industry from coal and other fuels and raw materials such as limestone and other additives. Mercury concentrations vary widely in fuels and raw materials and significant emission reductions can be achieved by switching to those with lower mercury content.

10.2.2 *Description of measure*

Process-oriented options are measures taken by producers to limit mercury emissions in various production processes including the control of point source pollution and the implementation of “good housekeeping” practices. Most of the relevant producers and users are covered by Directive IPPC. Measures include pre-treatment of coal, air pollution control technologies and mercury-specific control in coal combustion facilities. The air control technologies are described in Table 7.

Table 7: Description of air pollutant control technologies

Air pollutant control technologies	Description
Fabric filters (FF)	Consist of semi-permeable material in the form of bags or sleeves, which trap particles and are mounted in an airtight housing (baghouse), which is divided into a number of sections. Fabric filters are also used as a second stage in acid gas control systems (e.g. SO_2 control).
Electrostatic precipitators (ESP)	Use the principle of electrostatic attraction to remove entrained particles from the flue gases. They consist of rows of discharge electrodes (wires or thin metal rods), through which a high voltage is applied, and which are placed between parallel rows of metal plates to collect the charged particles.
Wet scrubber	Remove acid gases (e.g. HCl, HF and SO_2) by washing the flue gases in a reaction tower designed to provide high gas-liquid contact. In the first stage, the gases are cooled by water sprays, removing HCl, HF, some particulates and some heavy metals. In the second stage, calcium hydroxide or another suitable alkali is used to remove SO_2 and any remaining HCl. SO_2 reduction efficiency is > 90 per cent using lime/limestone wet scrubbing. Wet scrubbers are also sometimes employed with the prime objective of removing particles. The majority of flue-gas desulphurization (FGD) systems used worldwide are wet limestone scrubbers.
Semi-dry scrubbers/spray absorber systems (spray drying)	Make use of an alkaline reagent slurry (usually calcium hydroxide) which is introduced as a spray of fine droplets. The acid gases (e.g. SO_2) are absorbed into the aqueous phase on the surface of these droplets and neutralised; the slurry is dried by the heat of the flue gas to form a dry product, which is collected in an electrostatic precipitator or fabric filter. The SO_2 reduction efficiency is > 90 per cent
Dry injection systems	Involve the injection of an alkaline reagent (e.g. calcium hydroxide or sodium bicarbonate) as a fine, dry powder to remove and neutralise acid gases. The neutralised product is normally collected in a fabric filter.
Selective non-catalytic reduction (SNCR) and selective catalytic reduction (SCR)	Aim at reducing nitrogen oxides (NOx) in the flue gases. The SNCR process involves injection of ammonia or urea near the furnace. An SCR system is based on selective reactions with injected additives in the presence of a catalyst. The additives used are mostly ammonia (gaseous and in solution) but also urea. Emission reduction of NOx with SNCR can be limited (up to 50 per cent) but lies between 70 and 95 per cent for SCR.

Adsorption using activated carbon/activated lignite coke	Several different technologies have been developed for mercury, VOC and dioxin control. These systems can also be fairly effective at removing HCl and SO_2.

10.2.3 *Effectiveness*

The mercury removal efficiency of multi-pollutant reduction technologies (ESP installation) is estimated at >63%. The mercury removal efficiency of air pollution control (particle and sulphur removal in scrubbers) is estimated to be >93 %.

Activated Carbon Injection when used on a commercial scale in conjunction with a particle control device, e.g. ESP or fabric filter can achieve a significant reduction of mercury emissions. Reductions of more than 90 per cent are possible. Chemically treated carbons (e.g. brominated carbons) are more effective than conventional, untreated activated carbon when treating flue gases containing higher amounts of elemental mercury vapour.

Sulphur-impregnated carbon filters are the most effective mercury-specific controls with efficiencies in the range of 93 - 99 per cent.

Table 8: Examples of removal efficiencies for coal combustion

Emission control technology	Estimated Hg reduction (%)
Dry ESP	> 63
Fabric filter (FF)	> 93
FF+wet or dry scrubber+sorbent injection	> 98
Dry ESP + wet or dry scrubber + sorbent injection	> 98

10.2.4 *Costs*

Air pollution control devices may shift mercury from the flue gas to residues such as fly ash or flue-gas desulfurization (FGD) scrubber sludge. Assessments of total control costs should include the potential cost increases from managing these residues due to impacts from changes in the composition of the waste, including increased mercury content.

The annual total costs (i.e. annual capital- and O&M costs) for multi-pollutant reduction technologies range from 1.3 US$ 2008/MWhe for ESP installation with an estimated mercury removal of >63 % to 2- 5 USD$ 2008/MWh for more advanced air pollution control (particle and sulphur removal in scrubber) with a mercury removal efficiency estimated at >93 %.

The capital costs of abatement equipment at the Best Available Technique (BAT) level (particulate control + FGD) amount to less than 5 per cent of the total coal-fired power plant costs (without fuel costs). If the cost of coal is included in the cost of the plant, this share drops to 3 - 3.5 per cent. Capital costs are assumed based on a technology lifetime of 20 years and a 4 per cent discount rate.

The costs for purchasing and installing carbon (and sorbent) injection systems and monitoring equipment are in the range from $1.2 million to $6.2 million (US$2008) per power plant, significantly lower than for other types of air pollution control for particle matter, sulphur dioxide or nitrogen oxides. In comparison, the average investment (purchase and installation) cost of a wet scrubber for sulphur dioxide control is more than $86 million (US$2008) per boiler. For power plants the investment cost of installing fabric filters in addition to the sorbent injection systems ranged from $12.7 to $24.5 million (US$2008).

Using activated carbon injection to achieve 90 per cent mercury emission control costs from about $30,000 to less than $10,000 per pound of mercury removed (equal to $22,000 to $66,000/kg).

10.2.5 *Secondary environmental effects*

Pre-combustion measures such as coal washing are often performed to reduce the sulphur and ash content of the coal. A variable fraction of the mercury in the coal is also removed during this operation. Pre-combustion measures have been shown to reduce mercury emissions by an average of 30 per cent. Blending coal with low mercury coal will reduce emissions accordingly.

10.2.6 *Geographical and time scale of effects*

This measure is particularly relevant in Poland and Germany, where coal combustion for heating is widespread. Implementing the new technology will take longer in Poland due to the necessary investments in the companies and practice.

10.3 Measure 3: Waste management - controlled incineration

10.3.1 *Description of source*

Mercury can be present in household waste in highly variable concentrations in different countries depending on the occurrence of mercury in household products and the existence of dedicated systems for mercury collection or whether the products are disposed of in the regular waste stream. Common product groups which may contain mercury are batteries, thermometers and fluorescent light sources.

Large-scale waste incineration is not common in many countries. Other forms of waste management or lack of controlled waste-handling systems also give rise to mercury emissions via, e.g. degassing from landfills or uncontrolled burning of waste.

10.3.2 *Description of measure*

Since all the mercury emitted to the air as a result of waste incineration originates from the mercury contained in the waste, the most important primary measure is to reduce the input of mercury to the incinerator. This can be accomplished by sorting and removing mercury-containing products from the waste stream prior to incineration. Mercury control in waste incineration is thus closely related to the issue of mercury in household products and medical devices, a fraction of which may end up in waste streams.

Hazardous waste which may have high mercury content is usually burned either in special technology incinerators or in rotary kiln type furnaces. Special technology incinerators include very low technology drum, grate, or muffle type furnaces. Other technologies such as supercritical water oxidation and electric arc vitrification, which treat hazardous waste, can be included in this group, although they are not necessarily classified as “incineration” technologies.

The most important primary measure to reduce the mercury emissions to air is to prevent mercury entering the waste stream. This can be achieved by pre-treating the waste, separating and removing the materials containing mercury, before incineration.

10.3.3 *Effectiveness*

Reducing the emissions of mercury to air from waste incineration is heavily influenced by the mercury content in the waste, the waste burning capacity of the incinerator, the type of incinerator (mass burn excess air or modular starved air), the way in which it is operated (e.g. whether it includes heat recovery) and the abatement measures installed in the plant.

In waste incineration, the removal efficiency can be enhanced by adsorbing the mercury vapours from the combustion chamber using acid gas adsorbent material or other adsorbents and then removing the particle-phase mercury. The effectiveness of different mercury removal technologies for waste incinerators is presented in Table 9.

As can be seen, simple electrostatic precipitators sometimes only have very low mercury removal efficiencies. Wet scrubbers or spray absorbents using limestone for acid gas removal have efficiencies of 55 - 65 per cent and 44 - 52 per cent, respectively. To obtain a high removal efficiency of more than 90 per cent, special absorbents/adsorbents have to be added, most often activated carbon.

Table 9: Mercury removal efficiencies of flue gas cleaning systems for waste incinerators

Equipment	Temp. (ºC)	HgCl2, %	Hg(0), %	Overall, %
Electrostatic precipitator (ESP)	180	0-10	0-4	0-8
Wet scrubbers	65-70	70-80	0-10	55-65
Wet scrubbers with conditioning agent		90-95	20-30	76-82
Spray absorbers+ FF (limestone)	130	50-60	30-35	44-52
Spray absorbers+ FF (special absorbents added		90-95	80-90	87-94
Entrained flow absorbers + FF (special absorbents added)	130	90-95	80-90	87-94
Circulating fluidised bed + FF (special absorbents added)	130	90-99	80-95	87-94
ESP or FF + carbon injection				50->90
ESP or FF + polishing wet scrubber				85

10.3.4 *Costs*

For basic particle emission control (ESP or FF), total annual costs of 0.18 US$ per tonne waste were estimated in the EU funded project ESPREME. For combinations of particle control with scrubbers, optimised ESPs and activated carbon injections, the estimated annual costs are in the range 4 – 12 US$ per tonne waste. The co-control of mercury in these examples ranges from 5 - 10 per cent in the first case and from 80 to 99 per cent in the second example. The average costs of emission control techniques for medical waste incineration (pre-dedusting with an ESP, followed by lime and activated carbon injection and finally a fabric filter) were estimated by Visschedijk et al. (2006) to be around 25 US$ per tonne annual incinerated waste.

10.3.5 *Secondary environmental effects*

Secondary environmental effects can include the emissions' reduction of other hazardous substances of special concern to the Baltic Sea.

10.3.6 *Geographical and time scale of effects*

This measure is relevant for those countries introducing new technology for waste incineration and for those producing large amounts of wastewater sludge. For small countries with low amounts of sludge, this technology is more expensive and therefore difficult to implement.

10.3.7 *Political enforceability*

This measure should be well aligned with other environmental goals, for example, reducing waste, especially hazardous waste. As sludge is produced in rather high quantities, its management is crucial, such as, e.g. currently in the Baltic States, where there are no waste incineration plants.

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XI. Cadmium (Cd)

11.1 Measure 1: Improvement of BAT and revision of BREF document concerning industrial air abatement

11.1.1 Description of source

Currently air emission is the main route of cadmium input to the Baltic Sea catchment area with air deposition amounting to 15 tonnes per year, which contributes to approximately 20-40% of the estimated total yearly load (see annex 0 of the Guidance Document on Cadmium). Cadmium is emitted as a result of cadmium processing and as a by-product in industrial processes, such as power production facilities, non-ferrous and ferrous metals production, municipal waste processing, and metal processing. The energy sector in Poland is the predominant source of air emissions reported in BSR for the year 2009 at the level of 10 to15 tonnes. Other industries are reported to emit around 4 tonnes per year with the main input also coming from Poland. It concerns large and medium companies, predominantly the Industrial Emission Directive (IED) facilities for which BAT rules are required. By 2008 1257 IPPC installations were registered in the energy production sector in the Baltic Sea region. Large emissions according to PRTR are reported at around 2.2 tonnes (PRTR 2009).

11.1.2 Description of measure

Cadmium emissions to the air are related to dust emitted with flue gases (ENTEC, 2005) especially to fine particulate matter - PM2.5. The improvements from a technical point of view comprise dust reduction measures: electrostatic precipitators, fabric filters, wet scrubbers, and advanced measures which are process specific air abatement techniques. The particular application depends on the type of industrial process and type of fuel (see annex 0). For example for solid fuel burning power plants and industrial facilities of 5-600 MW capacity the emission reduction takes place through the installation of a fabric filter, for heavy oil fired boilers with 1-5 MW capacity an electrostatic precipitator (ESP) can be used, and for coal fired power plants with 300-1,300 MW capacity a 2- 3 stages ESP+ wet flue gas desulphurisation (FGD) system is suitable (Karvosenoja, 2007).

Industrial processes are constantly improved due to economical and technical changes. The IED (IPPC) Directive requires continuous improvement in the most environmentally important sectors and processes which are related to energy transformation, including metallurgy, the energy sector and waste incineration. Despite the current achievements (EC, 2008) there is still a potential for improvements and control of industrial processes which can be based on existing and emerging technical solutions.

11.1.3 Effectiveness

The advancement of compliance with the BAT approach and potential emission reduction depends on the baseline situation and specific local conditions of a particular facility. The biggest reduction potential lies in installations with no abatement or low efficiency measures such as multicyclones (50% efficiency). Additional possible reduction is as follows: for 5-50 MW energy installations using a 1-stage electrostatic precipitator ESP results in a reduction of 93%, using a fabric filter 99.7%, and applying a 2- stage electrostatic precipitator (ESP) 96 %. For large power plants a combination of 2- 3 stages ESP+ wet FGD a 99% efficiency is achieved (Karvosenoja, 2007).

11.1.4 Costs

Additional costs for advanced abatement techniques are between low to moderate in comparison to current air abatement technologies and depend on the technology used. These are accordingly: 380-480 euros per mg of PM2.5, 2-3 stages precipitator ESP+ wet desulphurisation FGD, 1-stage ESP 5-50 MW solid fuel 260-2300 euros per mg of dust, and for <50 MW power plants 220-13,000 euros per mg of dust. Fabric filter installation costs were reported in the range of 330-2900 euros per mg. It can be estimated that the cadmium reduction costs range from 0.7 to200 thousand euros per kg Cd) For industrial processes ESP/ESP + scrubber/fabric filter abatement costs range between 17 and 1,500 euros per mg of dust (Karvosenoja, 2007).

11.1.5 Secondary environmental effects

For advanced flue gas treatment techniques additional energy might be required. The need for wastewater treatment and disposal of hazardous waste should be taken into account. The abatement techniques also have a positive effect on total pollutant emission reduction including: PM10, PM 2.5, heavy metals and organics.

11.1.6 Technical feasibility

The technical feasibility of dust reduction measures is medium to high. Improvements in process technologies and end of pipe measures are in general feasible but facility specific conditions should be taken into account e.g. additional infrastructure might be required. Improved simple and advanced electrostatic precipitators, wet flue gas systems. fabric filters and cyclones for installations of 50 kW - 50 MW are well developed, to give some examples of what exists in the Baltic countries.

11.1.7 Secondary socio-economic effects (including indirect costs)

The additional costs for further environmental improvements are important especially for the sector of Small and Medium-sized Enterprises (SME) which can temporarily have a negative effect on competitiveness (additional costs) but at the same time it can have a positive effect on the economical performance (company image, better technical performance).

11.1.8 Geographical and time scale of effects

The measure is relevant for the following countries: Poland, Russia, Eastern Baltic Sea Region, Sweden, and Germany. The implementation timeframe is 5 (existing regime) to 15 years (new developments) depending on the BREF revision process, investment cycles in the companies and practice.

11.1.9 Political enforceability

Enforceability is medium (small and medium companies) and high (large companies) through regulations and existing industry policy control. The technical improvements are basically stimulated by appropriate EU regulations : the emission standards and the BAT approach with other EU policies including PM 2.5 requirements. Enforcement requires establishing new technical standards. The new IED regime (former IPPC) with foreseen improved environmental standards is an important factor (EC, 2010). Furthermore, newly considered air emission limit values (ELV) for dust in combustion processes and industrial processes in stationary sources foresee a further 50% reduction in 1-50 MW energy plants retrofitted with electrostatic precipitators, fabric filters, cyclone, and a 90 % reduction in small 50 KW to 1 MW power generation facilities.

Problems (institutional) in implementation can be expected on a country or regional level due to country relevance of industrial sectors. The role of public institutions is to further promote and build the capacity, an overall control system and evaluation. The enforcement can be strengthened by good implementation practice with relevant consultancies and administrative procedures, monitoring, treatment requirements and emission levels to the environment being defined. The measure refers also to voluntary agreements and management systems like ISO 14000 and EMAS securing good practice conduct and sound investment in environmental protection infrastructures.

11.1.10 *Cost-effectiveness analysis*

The highest reduction potential on the scale of the Baltic Sea catchment area is achieved in the case of air abatement measures at medium and smaller installations (<50 MW) with moderate and high cost-effectiveness. The potential is up to 50% in comparison with current standards being implemented – EU target level 2016. For example the marginal costs for solid fuel burning power plants and industrial facilities of 5-600 MW capacity are 2,200 to 3,300 euros per mg of dust (50-70 thousand euros per kg Cd), and for boilers with 1-5 MW capacity 6,900 Euros per mg of dust (Karvosenoja, 2007). 50% reduction of 10 tonnes per year would cost around 250 – 350 mln euros in the BSR region (Karvosenoja, 2007).

11.2 Measure 2: Improvement of BAT and revision of BREF document concerning wastewater treatment

11.2.1 *Description of source*

Direct and indirect discharges from industrial point sources give yearly between 3 and 7 tonnes of cadmium load to water in the BS catchment area. The IED directive is relevant for most of the industrial sources of PCDD/F. The baseline standard is defined by BREF documents setting the technological practices. Wastewater contaminated with cadmium and its compounds is generated in various technological processes in industrial sectors: especially in mining and ore processing (~ 1.3 tonnes in Poland), mineral and paper (~ 0.7 tonne in Poland, Finland, Sweden) non-ferrous and ferrous metal production (~ 0.3 tonne), the energy sector (0.1 tonne), recycling (0.1 tonne), and in metal industries, phosphate production and the chemical industry. It should be noted that small and medium companies with low emission discharge mostly through municipal WWTP and large companies have direct discharges of treated wastewaters to surface waters or sea water.

11.2.2 *Description of measure*

The amount of consumed water and cadmium occurrence in effluent and the achievable emission values differ for individual facilities. The improvements from a technical point of view comprise the following BAT approaches.

Optimization of technological processes for waste water treatment, e.g. basic wastewater treatment (pre-treatment, additional steps: pH/precipitation, coagulation, retention time adjustment, sand filtration, pre-treatment of wastewater from technological processes), processing of wastes from recycling facilities, metallurgy, steel production. For the predominant sources: mining, mineral and paper industries, effective clean up of large volumes and low cadmium concentrations with optimised sand filtration techniques and artificial ponds as polishing steps can be applied. For this purpose emerging absorption techniques can be useful in achieving high efficiency. These techniques can also be important for a reduction of the current costs of wastewater treatment

Advanced water abatement techniques as an additional step are applied mostly in special applications for technological process waters. These are: biological methods, ion exchange, sorption on modified active carbon, membrane filtration, nanofiltration, electrochemical techniques. They can improve the discharge quality e.g. in the metal industry or electronics. For that purpose the ion exchange provides especially flexible solutions.

11.2.3 *Effectiveness*

The potential effectiveness of improved treatment is between low and medium. The effectiveness of reduction at plant scale depends on the advancement in compliance with the BAT approach and site facility specific conditions. It depends on the ways to reduce the wastewater volume, recycling of water and the effluent treatment type: in a local or a central wastewater treatment plant. In proper conditions precipitation as the baseline technique has an efficiency of 99 –99.9%. Cd concentrations in the nonferrous and ferrous industry after treatment can be achieved as low as <0.05 mg/l using the precipitation technique with an efficient coagulation process (Rao, 2010, BREF). The electrocoagulation removal for Cd in process water is 95-99 % (0.3 mg Cd /l treated). Costs for wastewater with low Cd levels < 5 mg/l are relatively high. On the other hand, very high concentrations require on additional treatment steps.

11.2.4 Costs

Costs are between low to moderate depending on the particular facility, the technology used and the type of abatement. Additional costs for optimised abatement techniques and additional simple techniques are moderate (5-15 %) in comparison to the current cost of wastewater treatment. Filtration costs are relatively low. The cost for electocoagulation and electroflotation treatment of cadmium and nickel from electroplating rinse water with 99.4% efficiency for cadmium removal is 0.05 euro per m3 (20 mg of Cd) to 0.7 euro/m3 for cadmium (Butler 2011). Costs for treatment of high wastewater volumes with advanced techniques are relatively high. Low-cost sorption materials can be an important factor in achieving low costs and high removal.

11.2.5 Secondary environmental effects

In optimisation of treatment performance the effects are in general positive. For the advanced water treatment techniques additional energy might be required and hazardous wastes generated - for electrocoagulation for example an energy use of 0.68 kWh, per m3 of wastewater is required. Improvements in technological processes and abatement techniques have a potential effect on the total pollutant emission reduction including emission of: PM10, PM 2.5, heavy metals, organics, low generation of wastes and wastewater.

11.2.6 Technical feasibility

Table 6: Cadmium-relevant water emission reduction techniques (SOCOPSE, 2009)

	Mitigation measure	Phosphates fertilisers production ()	Non ferrous industry (BREF)	Ferrous industry (BREF)
Optimisation	Baseline techniques			
	Precipitation	X	W	X
	Flocculation/flotation	X	X	X
	Sedimentation	X	X	X
	Filtration	X	X	X
	Neutralisation	X	X	X
Selection	Additional measures			
	Ultrafiltration/nanofiltration	X	X	X
	Reverse osmosis	X	X	X
	Ion exchange	X	S	S
	Activated carbon	X	X	X
	Electrotechnical	P	P	P

W widely used, S specific problems/situations, P – potential

Sources: BREFs

The overall technical feasibility is between medium and high. Improvements in process technologies and end of pipe measures are feasible in general but facility-specific conditions should be taken into account. Optimisation of technologies relies on existing solutions tailored to particular needs. One can distinguish between (BREFs) main technological approaches such as pre-treatment, drainage system design, management of raw materials , closed water cycles , recycling and re-use, pre-treatment of waste water from technological sources, low level cadmium in raw materials, optimisation of basic wastewater treatment, and combinations of advanced techniques such as: ion exchange, sorption on active carbon, membrane filtration, nanofiltration, electrochemical techniques. Barriers of implementation for particular techniques might be important in specific technological conditions.

The development of new technologies depends on the specific issues arising within the production processes. New technologies are not required for general processes.

11.2.7 Secondary socio-economic effects (including indirect costs)

The additional costs for further environmental improvement are important especially for the sector of Small and Medium-sized Enterprises (SME) which can temporarily have a negative effect on competitiveness but at the same time under specific conditions it can have a positive effect on the economical performance.

11.2.8 Geographical and time scale of effects

The measure is relevant especially for Poland. Zinc and lead ore mining facilities in Poland (Zakłady Górniczo-Hutnicze “Bolesław”) are reported to emit around 0.9 tonne. It is less relevant for Finland, Russia, Sweden, and Germany. The timeframe for implementation is 5-10 years depending on the investment cycles of the companies. A future decline in emissions should be expected due to the reduction of cadmium use in industrial processes.

11.2.9 Political enforceability

Political enforceability is medium (small and medium companies) to high (large companies) through regulations and existing industry policy control. The technical improvements are in principle stimulated by appropriate regulations which are of European dimension. These are: IED, industrial wastewater discharge limits, landfill effluents quality standards and treatment regulations. For example the required effluent standard for the galvanic industry is <0.1 mg of Cd/l. Problems (institutional) for efficient implementation, monitoring and control can be expected on a country or regional level.

11.2.10 Cost-effectiveness analysis

For the whole catchment area the reduction of wastewater discharge is also important and moderately cost-effective. The industrial has a 10% share of input in the catchment area (3 tonnes). A potential 70% reduction of concentration in effluents (end of pipe measures) in comparison to current emission reduction standard in effluents can be assumed (see annex 0 of the Guidance Document on Cadmium) An additional elimination of 2 tonnes of Cd in discharge to water would cost roughly around 10,000 mln euros (5 mln per kg Cd removed).

11.3 Measure 3: Replacement and Retro-Fit of household heating furnaces

11.3.1 Description of source

Air emission from the residential sector is important for cadmium deposition in the catchment area and directly into the Baltic Sea. It concerns individual households with 10-100 kW heat power boilers, both in detached houses and apartment buildings, medium-size energy plants 100 kW –1MW for SMEs and central heating plants > 2 MW owned by private and public entities. Low efficiency of the burners, low fuel quality and waste burning cause emissions of particulate matter containing Cd.

Around 26 tonnes are reported to be emitted in the region, predominantly in Poland. It constitutes around 30 to 40% of the total load in the Baltic Sea catchment area with air deposition.

11.3.2 Description of measure

The emission reduction is based on the retrofitting and modernisation of boilers, housing infrastructures like central heating systems, flue gas treatment utilities, and a switch to high quality fuel. There is a variety of technical solutions related to solid fuels concerning burning installations: secondary combustion chamber, improved conditions in the afterburner chamber, catalysts for wood burners, installation of an accumulator tank, and a clean-up of flue gases using electrostatic precipitators - ESP. The development of district heating and thermal insulation of buildings is an important factor for the final efficiency and technique selection .

11.3.3 Effectiveness

The efficiency of technical dust emission reduction measures on a household level is high - up to 50 to 99% reduction depending on technical solutions and current technical standards applied but in practice it depends also on the awareness and conduct of house owners. For example automatic boilers with good fuel have a 95 to 99% reduction efficiency, and for electrostatic precipitators for domestic wood combustion it is 80-95%.

11.3.4 Costs

Costs vary between medium to high according to technical solutions and facility conditions. They are incurred by house owners or public entities. Investment costs for new installations with a high emission dust reduction are low to medium (1000 - 2000 Euros) in comparison with the baseline investment in the simplest furnaces with a low dust emission rate (400-500 euros). Additional costs for auxiliary installations, montage and service should be included. Operational costs for house owners depend on the maintenance and fuel costs. The high quality fuel including high grade coal is 50% more expensive compared to the commonly used lowest grade coal (culm 200 euros per tonne – see annex 0 of the Guidance Document on Cadmium) . For small ESP with 80-95% efficiency the investment costs are 420 to 15,000 euros per mg of PM2.5 (Karvosenoja, 2007) – around 8 to 300 thousand euros per kg Cd depending on the efficiency. For the replacement of boilers in Poland the roughly estimated costs for a PM10 emission reduction are around 3,000 to8,000 euros per mg of dust (60 to160 thousand euros per kg of Cd).

11.3.5 Secondary environmental effects

In case of dust removal additional waste is generated and for automatic boilers energy is used. The energy demand for producing different fuels differs substantially. Positive aspects of the measure are PM2.5, PM10, GHG reduction, waste production by consumers and improved health conditions.

11.3.6 Technical feasibility

Apart from the ESP measures the installations are commercially available and with many competing producers the market is well developed. There is also potential in all BSR countries for producing high quality fuels. Reduction of residential combustion emissions can be achieved through replacement/improvement of individual combustion installations:

- Improvement of combustion processes. Solutions for households and small district heating plants are available.
- Efficient small burners for high quality fuels: commonly available automatic high-efficiency systems

- Technical measures for air emission reduction: dust filters feasible in medium and small installations. Currently this approach is rarely used e.g. in Germany.

11.3.7 Secondary socio-economic effects (including indirect costs)

The cost of heating is an important share of the household expense in Poland and other EBSR countries. A positive aspect is the reduction of health risks.

11.3.8 Geographical and time scale of effects

The issue is most relevant for Poland (82 % of the source emission), and Russia (13%), less important in Estonia, Latvia (1.2 %), and Lithuania (see annex 0 of the Guidance Document on Cadmium). It is important on a local level. The time frame of these measures taking full effect varies between 10 and 30 years including the establishment of good practices and investments. The time frame would depend on the public support in terms of implementation of policies.

11.3.9 Political enforceability

The measure can be implemented on a voluntary basis. The enforceability depends on the country specific situation, in Poland it is medium. It is crucial to provide public support to house owners, public entities and entrepreneurs in improving energy efficiency and the environmental performance of households. In Poland financial support examples exist but a more dynamic approach is difficult to implement as the problem is not sufficiently recognized by the public and decision makers. Incentives can be provided either in form of tax deductions, credits, loans or subsidies.

11.3.10 Cost-effectiveness analysis

Cost effectiveness varies between low and medium. The reduction potential is region- and problem-specific. A 15% reduction of air deposition in the catchment area is realistic. For Polish cities a 50% reduction between 2006 and 2020 is possible. Marginal costs for an electrostatic precipitator (ESP) scenario are 420 to 3,700 mg of dust corresponding to 8 to 70 thousand euros per kg of Cd. A reduction of 6 mg of Cd deposition in the BS catchment area would cost 48 to 400 mln euros. For replacements of boilers the cost would be around 360 mln euros.

11.4 Measure 4: Waste management – recycling

11.4.1 Description of source

The emission is caused by inadequate disposal of products and materials containing intentionally used cadmium and its compounds or as impurity to other materials. Waste is generated during production, consumption and disposal of the products. Uncontrolled emissions arise from neglect or low performance in collection, waste handling, and disposal, processing of wastes and recycling. It is the result of improper behaviour and practices of consumers, SMEs and recycling companies and insufficient hazardous waste management schemes in the industries. For some applications the recovery rate is high e.g. 95% in PL of recycling of Cd-Ni batteries used in the energy sector (GUS, 2010). Despite the efforts the current recovery rate can be improved. In principle the use of cadmium in products today is restricted to special applications, and not intended for general consumption. Nevertheless, it is still used in some applications like hearing aids and in special applications in the industrial sector, for example in batteries. The disposal of ashes from solid fuel burners is also an important issue. In the years 2007 to2009 1910 tonnes of Cd were used in products in the 27 EU countries and 273 tonnes (jewellery) were imported (RPA, 2010). It is also present as an impurity in some materials e.g. construction elements made of zinc (Table 7). Cadmium is present and will be present in waste streams for the next 10 to 20 years after the implementation of the ban of cadmium use. Furthermore, according to new directives it will remain in the products in recycled materials.

Table 7: Use of products containing cadmium

Type of use
Conductive and electroplating agents, solders
Batteries and cells - consumers[11]
Industrial batteries and cells - cordless power tools[12], medical equipment, power supply
Colouring and reprographic agents fillers
Heat stabilisers in flexible PVC

In principle a municipal waste stream is disposed in controlled landfills or incinerated and leachates from landfills are in most cases treated in MWWTP and end up in sewage sludge (see chapter 6.8).

11.4.2 Description of measure

The measure focuses on implementing and/or strengthening good practices in organisational, consumer and administrative practices concerning product dismantling, waste collection, processing, and its disposal. It can be achieved through promotion of:

- Efficient local level management schemes of municipal and hazardous waste collection, organisation and disposal.
- Improved processing and use of recycled material e.g. construction materials – based on limits for materials containing cadmium or special re-use requirements.
- Good practices in the separate collection of products and disposal schemes on a country level: e.g. marketing schemes including return of the used appliances. Good example is the voluntary plastic industry PCV recovery scheme.

11.4.3 Effectiveness

Effectiveness of well operating waste management systems should be high in principle taking into account the elimination of cadmium input to the environment in comparison with an improper disposal of products and consecutive soil and water contamination.

11.4.4 Costs

Additional costs are comparatively low taking the existing standards of waste collection and disposal as a baseline. They comprise improvements in collection systems, either organised by municipalities or by relevant branches e.g. automotive, electric. For the recycling companies' investment and operational costs for product dismantling lines should be included. Additional costs have to be incurred by SMEs for organising their waste management and by consumers for transport. For the public sector costs for information and promotion campaigns should be included.

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[11] DIRECTIVE 2006/66/EC OF THE EUROPEAN PARLIAMENT AND OF THE COUNCIL of 6 September 2006 on batteries and accumulators and waste batteries and accumulators and repealing Directive 91/157/EEC (Text with EEA relevance)

[12] RoHs Directive restrictions on use in cordless tools.

11.4.5 *Secondary environmental effects*

The measure is important from the perspective of sustainability – material recovery, lower landfilling rate and less impact of other substances on the environment. Additional energy use is required for improved logistics schemes.

11.4.6 *Technical feasibility*

There are no special technical requirements. Organisational practices and skills with standard technical solutions are the most crucial in improving the waste management schemes. The cultural, administrative and organisational barriers are potentially important in implementation.

11.4.7 *Secondary socio-economic effects (including indirect costs)*

Potential rise in prices of selected products or costs of waste management for citizens can be expected. Additional costs for SMEs for reorganising their waste handling can be an additional burden.

11.4.8 *Geographical and time scale of effects*

The measure is most relevant for Poland and the BSR but also for other countries. The time for improving or establishing efficient systems can be estimated at 2 to 5 years. This measure is important as much of the product stock is in use and has to be recycled in 5 to10 years.

11.4.9 *Political enforceability*

It is based on existing waste product recovery requirements. Legal obligations for safe disposal of electronic equipment and automotive waste posed on consumers and retailers are already in place. A set of directives is already implemented (see annex 0 of the Guidance Document on Cadmium)

11.4.10 *Cost-effectiveness analysis*

Cost effectiveness is low to medium taking into account the potential of cadmium input in the catchment area (1 tonne of Cd input to land assuming 1% of improper disposal). From the Baltic Sea perspective it can be even less important as the direct input from this source is low. From that perspective it is the most cost-efficient in the coastal areas reducing the load with run-off water and residual waste in the environment. It is important for villages and towns and service areas without proper collection and selection of wastes and poor product dismantling and waste processing practices.

11.5 Measure 5: Reduction of Cd content in fertilisers

11.5.1 *Description of source*

Use of phosphorous fertilisers or sewage sludge and compost contaminated with cadmium in agriculture or in amelioration of none-agricultural soils is a source of cadmium emission to soil. Taking into account a cadmium content of 8 mg / kg fertiliser 3 tonnes of cadmium are emitted to soil in Poland every year – around 5 tonnes in total in the BSR.

11.5.2 *Description of measure*

The measure focuses on the improvement of legal requirements already applied on an EU and country level coupled with a code of conduct respected on the farm and enterprise level. These comprise the following soil amelioration requirements:

- Stricter limit on Cd content in phosphorous fertilisers, sewage sludge and compost applied in agriculture and in soil amelioration.
- Ban/regional/conditional restrictions on the application of sewage sludge and compost.

11.5.3 Effectiveness

The potential for reduction of Cd input to land in the Baltic Sea catchment area is relatively high. Taking into account the difference between high (8 mg) and low content (2 mg Cd/kg of P2O5)in phosphorous fertilisers and other materials the flux to agricultural and municipal soils can be reduced at least by 50% in comparison with the current situation. On the other hand the direct input to surface water and the Baltic Sea is low as cadmium is contained in soils. Thus, this source is important only for long term and local perspectives.

11.5.4 Costs

The costs of Cd reduction depend on many factors. In general there are no cost-efficient purification technologies for phosphorous fertilisers available and availability of Cd low content raw material on the global market is limited. Additional processing of raw materials and use of low cadmium raw materials can lead to an increase in prices of fertilisers (Oosterhuis et al., 2002). Restricting sewage sludge use in soil amelioration can also have an impact on using sewage sludge and compost as cost efficient alternatives for fertilisers.

11.5.5 Secondary environmental effects

The processing of fertiliser raw material has an additional environmental impact. Lower input to soil of other persistent contaminants with sewage sludge and compost can be expected. Better use of organic material in agriculture and better maintenance of soils give an additional benefit.

11.5.6 Technical feasibility

The cadmium reduction in phosphate fertilisers implies a number of choices presented in annex 0 of the Guidance Document on Cadmium. For other non-technical measures there are no barriers. A new source of phosphorous, e.g. wastewater treatment plants, can be considered as low cadmium sustainable solutions.

11.5.7 Secondary socio-economic effects (including indirect costs)

For farmers higher costs of fertilisers can be expected. Additional costs are connected to organic matter processing and use of fertilisers of Cd low content. Limiting Cd content in phosphorous fertilisers can raise the costs of farming.

11.5.8 Geographical and time scale of effects

It is relevant for Poland, Latvia and less for other countries. The time frame is around 10 years – establishing good practices or changing the fertiliser market through regulations.

11.5.9 Political enforceability

The enforceability is between low and medium. It depends on agricultural policies and socio-economical conditions of the agricultural sector. The EU level is the most appropriate to tackle the issue. The implementation can be strengthened by code of conduct in organic farming: use of compost and improved monitoring of sewage sludge quality.

11.5.10 Cost-effectiveness analysis

The cost-efficiency of fertiliser purification is low. The replacement of raw material can also not be efficient as the availability of low content raw material is limited. The most cost-efficient (medium to high) is the replacement of high Cd content fertilisers with organic matter and an efficient and environmentally sound handling and application of sewage sludge (restrictions and control). These approaches might be very effective in the long term.

11.6 Measure 6: Advanced waste water treatment

11.6.1 *Description of source*

Cadmium is effectively removed at municipal sewage treatment plants (>90%) in activated sludge processes of biological treatment. The achieved concentrations in effluents are in most cases in the range of EQS for cadmium. Despite this in Poland and the East Baltic Sea Region the infrastructure needs upgrading to achieve the best performance parameters. A high level of reduction is achieved in around 80% of MWWTP in Poland. Around 1 tonne of Cd is discharged every year inland and 0.7 tonne to coastal waters from operating MWWTP (COHIBA estimations). Additionally the untreated wastewater discharges (40% in Poland) and run-off management should be also taken into account. The reduction potential is around 5 % of the total load to the Baltic Sea catchment area determined for agglomerations with high industrial discharge and rural areas.

11.6.2 *Description of measure*

The removal of cadmium from the waste water is achieved by application of the appropriate combination of a wide range of physical, chemical and biochemical processes. A selection of technical solutions is specific to problem areas with the following options:

- infrastructure for wastewater treatment in small and medium MWWTP and individual houses in rural areas,
- simple clean up techniques of storm water in industrialised and highly urbanised areas e.g. sedimentation and buffer ponds, filtration beds,
- additional measures for MWWTP (tertiary): coagulation and pH adjustment, filtration, sedimentation as the basic approach and advanced techniques as ultrafiltration.

11.6.3 *Effectiveness*

Currently, the achievable levels of emission reduction are around 90-95% of suspended matter along with Cd in effluents in the three step treatment systems. Biological methods are very efficient in cadmium removal:>99%. For dissolved cadmium the ion exchange, sorption on active carbon, membrane filtration and nanofiltration additional reduction is low. The most effective is an upgrade of older MWWTP, infrastructure in rural areas and storm water management in industrial areas. Prolonged retention time and additional sedimentation are an efficient approach.

11.6.4 *Costs*

Costs depend on particular facility conditions. For sand filtration the costs include around 50 euros/m3 of investment costs and 1-2 euros/m3 operational costs. This amounts to roughly around 40,000 euros per kg cadmium removed. For new investments the costs are 300,000 euros per kg of Cd removed (based on COHIBA estimations). For MWWTP standard costs of upgrading the tertiary treatment e.g. membrane technologies, range from 100 euros/m3 investment costs and 3-4 euros/m3 of operational costs for small units with daily flow of 50 thousand m3.

11.6.5 *Secondary environmental effects*

Additional waste and electricity use should be taken into account. The recovered water can be used for example in agriculture.

11.6.6 *Technical feasibility*

Technical feasibility is medium to high. The basic technologies are mature and available. It concerns also small units designed for rural areas and run-off management techniques but their application depends on site-specific conditions. Improvement are observed in the performance of advanced techniques and costs:

ion exchange, sorption on active carbon, membrane filtration, nanofiltration in MWWTP.

11.6.7 Secondary socio-economic effects (including indirect costs)

The improvements can result in an increase of water prices but also provides opportunities of using the treated water.

11.6.8 Political enforceability

Most of the requirements are already in place (EU Water Framework Directive). The WFD has a strong impact on the implementation of the basic treatment requirements for settlements with more than 2,000 inhabitants. For further improvements other benefits need to be implemented.

11.6.9 Cost-effectiveness analysis

Cost-effectiveness is medium with a 5 % reduction of the current load to the Baltic Sea catchment area. The cost-effectiveness of measures for advanced treatment is low (30 mln euros for 100 kg) in terms of sole cadmium removal which should be taken into account as a co-benefit. In Poland around 40% of municipal wastewater is not treated in MWWTP and investments in standard treatment can reduce cadmium load to environment in a cost-efficient manner by 0.1 – 0.3 tonnes (4 mln euros).

11.7 Measure 7: public awareness raising

11.7.1 Description of source

According to the relevant regulations, a substitution in products - batteries and cells in general use and stabilisers in PCV - is almost completed in Europe. By 2012 the use of cadmium in products of general consumption will be eliminated (jewellery, electronics, batteries). Cadmium is used only in special applications in relatively low quantities. These are:

- Paints colouring agent still in use
- Recycled PCV
- Solar panels
- Electroplated materials still in use but with restrictions
- Coated materials e.g. porcelain
- Metal parts coated with Cd

A use of products containing cadmium can lead to direct emissions to water and soil. Although the reduction potential for the whole catchment area is low it can be important locally for agglomerations discharging the effluents directly to the Baltic Sea.

11.7.2 Description of measure

Awareness campaigns focusing on relevant stakeholders (product users) on buying for example imported products, proper use of products or product substitution can limit the emissions in urban areas. The measure is addressed to special groups e.g. artists, small and medium companies and workshops dealing with materials containing cadmium.

11.7.3 Effectiveness

The effectiveness is high on a user scale depending on substitution – at a rate of 100% or with a careful handling approach 70-90 %, but it can be below 10% to a medium 50% for the broad range of users.

11.7.4 Costs

Costs will be incurred for awareness campaign activities and training of entrepreneurs or selected user groups. Costs are between low to moderate for information and awareness campaigns and trainings. In a Swedish case, where a campaign for raising awareness was carried out in the Stockholm area targeting painters and art schools, a reduction of 1 kg Cd in the sludge cost 5,000 euros.

11.7.5 Secondary environmental effects

Potential emission reduction of other contaminants or other benefits as lower use of water, lower waste production.

11.7.6 Geographical and time scale of effects

All Baltic Sea region countries but with special focus on urban areas. The time frame is 3-10 years depending on the population or target group.

11.7.7 Political enforceability

The actions should be undertaken by local administration or NGOs. The measure requires a good recognition of the problems on local level. From that point the enforceability is between medium to high.

11.7.8 Cost-effectiveness analysis

It should be underlined that the amount of cadmium discharges from these sources is relatively low and very specific to certain locations. Thus, the cost-effectiveness is expected to be medium.

11.8 Measure 8: Treatment of contaminated soil

11.8.1 Description of source

Water flow from naturally and anthropogenically contaminated soil, surface water and sediments is a substantial but still poorly recognised source of cadmium discharges in the Baltic Sea catchment area. General contamination in urban and industrial areas as a result of run-off from impervious surfaces is also an important source.

11.8.2 Description of measure

A variety of techniques for soil remediation exist. Phytoremediation and phytostabilisation are important for large areas with relatively low concentrations. For former waste landfills the natural evotranspiration cover is an important approach.

11.8.3 Effectiveness

The effectiveness of emission reduction depends on the particular site potential of leaching cadmium. For medium heavy metal contaminated sites the leaching reduction can be high on a local scale but moderate on a catchment area scale: 10% reduction.

11.8.4 Costs

Costs depend on the particular contamination situation on site and the technology used.

11.8.5 Technical feasibility

Technical feasibility depends on the particular problem.

11.8.6 Secondary socio-economic effects (including indirect costs)

Positive effects can be expected in other sectors. Remediation efforts should be coupled with land reclamation thus reducing the costs and burden on local communities.

11.8.7 Geographical and time scale of effects

The issue is most important for Poland, Russia, Lithuania, Latvia and Eastern Europe. The time scale of effects is over 10 to30 years from a Baltic Sea perspective.

11.8.8 Political enforceability

Political enforceability depends on the local situation. The prerequisite is the national implementation of contaminated site management rules including the identification of sites. It varies between low and high.

11.8.9 Cost-effectiveness analysis

It should be underlined that the amount of cadmium discharges from these sources is relatively low and very specific to certain locations. It can be cost effective on a local scale for distinct hot spots. But generally the cost-effectiveness is expected to be low with reference to only the cadmium reduction.

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