The National Swedish Contaminant Monitoring Programme for Freshwater Biota, 2016

Övervakning av metaller och organiska miljögifter i limnisk biota, 2016

Elisabeth Nyberg, Suzanne Faxneld, Sara Danielsson, Anders Bignert,

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Swedish Museum of Natural History
Department of Environmental Research and Monitoring
P.O. Box 50 007
104 05 Stockholm
Sweden
The National Swedish Contaminant Monitoring Programme for Freshwater Biota, 2016

Preparation of samples and biological parameters:
Swedish Museum of Natural History
Henrik Dahlgren, Douglas Jones, Eva Kylberg, Jill Staveley Öhlund, Mikael Anderssson

Chemical analysis and review of the chapters connected to the specific compound:

Organochlorines:
Department of Environmental Science and Analytical Chemistry, Stockholm University
Project leader: Cynthia de Wit
Chemists: Ulla Eriksson, Anna-Lena Egebäck

Perfluorinated substances:
Department of Environmental Science and Analytical Chemistry, Stockholm University
Project leader: Jon Benskin
Chemists: Merle Plassman, Raed Awad

Trace metals:
Department of Environmental Science and Analytical Chemistry, Stockholm University
Project leader: Marcus Sundbom
Chemists: Pär Hjelmquist, Frida Edberg

PCDD/PCDF:
Department of Chemistry, Umeå University
Project leader: Peter Haglund
Chemist: Peter Haglund

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Report authors
Elisabeth Nyberg, Suzanne Faxneld, Sara Danielsson, Anders Bignert
The Department of Environmental Research and Monitoring, Swedish Museum of Natural History

Responsibe publisher
Swedish Museum of Natural History

Postal address
Naturhistoriska riksmuseet
Box 50007
104 05 Stockholm

Telephone
+46(0)8-519 540 00

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Summary
The report summarises the monitoring activities within the National Swedish Contaminant Programme for freshwater biota. Each monitored contaminant has been examined in pike, perch or Arctic char from 32 lakes from the north to the south in Sweden.

No general trend could be seen for mercury, and all samples, except Arctic char from Abiskojaure, were above the EU target level. Lead was generally decreasing. Nickel showed a general upward trend in perch. An increase, or indication of an increase, was seen for chromium during the last ten years. Zinc concentrations were decreasing in a majority of the perch samples and in pike from Lake Storvindeln. Arsenic concentrations were increasing in Arctic char and pike from all sites and in perch from a third of the sampling sites. No general trend was observed for copper, silver, aluminum, tin, bismuth, and cadmium.

In general, downward trends were seen for; PCBs, HCHs, DDT, DDE and PCDD/Fs in all species (with a few exceptions). The chlorinated compounds generally show a somewhat higher concentration in the southern parts of Sweden. Most chlorinated compounds were below the suggested target levels.

No general trend was seen during the monitoring period for PBDEs. Concentrations of PBDEs in Lake Bolmen increased from the 60s and peaked in the late 80s to mid-90s and have subsequently decreased. The concentration of HBCDD was under LOQ in a majority of the samples.

Several of the perfluorinated carboxylate compounds showed an increase in concentration in perch, while PFOS and FOSA decreased. Higher concentrations of perfluorinated compounds can in general be seen in the southern part of Sweden. In all lakes, PFOS is below the target level for all species.
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### 11 BIOLOGICAL VARIABLES
1 Introduction

This report summarises the monitoring activities within the National Swedish Contaminant Monitoring Programme for freshwater biota. It is the result of joint efforts from the Department of Applied Environmental Science at Stockholm University (analyses of organochlorines, flame retardants, perfluorinated compounds and trace metals); the Department of Chemistry at Umeå University (analyses of PCDD/PCDF); and the Department of Environmental Research and Monitoring at the Swedish Museum of Natural History (co-ordination, sample collection, administration and preparation, recording of biological variables, freeze-storage of biological tissues in the Environmental Specimen Bank (ESB) for retrospective studies, data preparation and statistical analyses). The monitoring programme is financed by the Environmental Protection Agency (EPA), Sweden.

The data in this report represents the bioavailable portion of the investigated contaminants i.e. the portion that has passed through biological membranes and may cause toxic effects. The objectives of the freshwater monitoring programme can be summarised as follows:

- to estimate the levels and normal variation of various contaminants in freshwater biota from representative sites throughout the country, uninfluenced by local sources;
- to describe the general contaminant load and to supply reference values for regional and local monitoring programmes;
- to monitor long term time trends and to estimate the rate of changes found; quantified objective: to detect an annual change of 10% within a time period of 10 years with a power of 80% at a significance level of 5%.
- to estimate the response in biota to actions taken to reduce the discharge of various contaminants; quantified objective: to detect a 50% decrease within a time period of 10 years with a power of 80% at a significance level of 5%.
- to detect incidents of regional influence or widespread incidents of ‘Chernobyl’-character and to act as watchdog monitoring to detect renewed usage of banned contaminants; quantified objective: to detect an increase of 200% a single year with a power of 80% at a significance level of 5%.
- to indicate large scale spatial differences; quantified objective: to detect differences of a factor 2 between sites with a power of 80% at a significance level of 5%.
- to explore the developmental and regional differences in the composition and pattern of e.g., polychlorinated biphenyls (PCBs), hexachlorocyclohexane (HCHs), dichlorodiphenyltrichloroethane (DDT), Dichlorodiphenylchloroethylene (DDE) polychlorinated dibenzo-p-dioxins and -furans (PCDD/F), polybrominated diphenyl ethers (PBDEs), hexabromocyclododecane (HBCDD), perfluorinated alkyl substances (PFASs), and various metals;
- the measured concentrations are relevant for human consumption as the species sampled are important for recreational fishing and are commonly consumed;
• all analysed, and a large number of additional specimens, of the annually systematically collected material are stored frozen in the Environmental Specimen Bank (ESB). This material enables future retrospective studies of contaminants unknown or impossible to analyse today, as well as control analyses for suspected analytical errors;

• although the programme is focused on contaminant concentration in biota, the development of biological variables e.g., length, age and fat content, are monitored at all sites.

• some of the monitored lakes are chosen because of additional investigations of water chemistry and fish population carried out by the Swedish University of Agricultural Sciences (SLU) and the Swedish Board of Fisheries respectively. These lakes still fulfil the original selection criteria (see chapter 6).

• experience from the national programme with time series of >30 years can be used in the design of regional and local monitoring programmes;

• the unique material of high quality and long time series is further used to explore relationships between biological variables and contaminant concentrations in various tissues; the effects of changes in sampling strategy, the estimates of variance components and the influence on the concept of power etc.;

• the accessibility of high quality data collected and analysed in a consistent manner is an indispensable prerequisite to evaluate the validity of hypotheses and models concerning the fate and distribution of various contaminants. It could furthermore be used as input of ‘real’ data in model building activities concerning freshwater ecosystems;

• by using target levels criteria, the results from the investigations can be used as a tool to prioritize pollutants and to find localities where there is a risk for effects on biota.

The current report displays the time series of analysed contaminants in biota, and summarises the results from the statistical treatment. It does not in general give background or explanations to significant changes found in the time series. Increasing concentrations thus require intensified studies. Short comments are given for temporal trends and spatial variation. However, it should be stressed that geographical differences may not reflect anthropogenic influence, but may be due to factors such as productivity, temperature, pH etc.

One of the 16 national goals for the Swedish environment is an environment free of pollutants. The definition of this goal can be translated roughly as follows:

The environment shall be free from substances and metals that have been created or extracted by society and that can threaten human health or biological diversity.

The national monitoring programmes are a part of this aim and the results are important in the follow up work.
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2 Summary

The environmental contaminants examined in this report can be classified into four groups – trace metals, chlorinated compounds, brominated flame retardants and perfluoroalkyl substances. Each of these contaminants has been examined in pike, perch and Arctic char from 32 lakes geographically spread from the north to the south of Sweden. The following summary examines overall trends, spatial and temporal, for the four groups.

Fat Content, Age and Length
Pike and perch displayed a downward trend in fat content at 75 % of the sites examined. However, an upward trend in fat content could be seen for Arctic char from Lake Tjulträsk for the whole monitoring period and in perch from Lake Skärgölen the last ten years. The age of the perch sampled within the programme was somewhat lower in the most southern and south eastern parts of Sweden, whereas the length of the perch was homogenous in all lakes sampled.

Trace Metals
No general temporal trend could be observed for mercury in the freshwater environment. However, in all lakes and species the concentrations are above the suggested EU target level of 20 ng/g wet weight.

Lead is generally decreasing over the study period (in time series of sufficient length), supposedly due to the elimination of lead in gasoline. In all lakes, lead concentration is below the suggested EU target level of 1.0 µg/g wet weight.

Cadmium concentrations show no consistent trends over the monitored period. It is worth noting that despite several measures taken to reduce discharges of cadmium, the most recent concentrations in Arctic char and pike are similar to concentrations measured 30 years ago in the longer time series. In about 30% of the lakes, the cadmium concentrations in perch are above the suggested EU target level of 0.16 µg/g wet weight.

Nickel concentrations varied a lot between sampling site and matix, but about half of the time series in perch showed an upward trend.

Chromium concentrations showed a general downward trend in all matrices during the monitoring period, but this decrease is most probably caused by the change of method for chromium analysis in 2004. However, when only the last ten years are considered, the majority of time series showed an increase – or indication thereof, in the chromium concentration.

The concentrations of zinc in perch liver are similar in all lakes monitored. The concentrations are decreasing significantly at a majority of the perch sampling sites and in pike from Lake Storvindeln.

Arsenic concentrations were increasing in liver from Arctic char, pike and perch from 30% of the sampling sites.

No general temporal trend were observed for copper, silver, aluminium, tin, and bismuth concentrations in fish liver during the monitoring period.
**Chlorinated Compounds**

Generally, a downward trend was observed for all compounds i.e. polychlorinated biphenyls (PCBs), hexachlorocyclohexane (HCHs), dichlorodiphenyltrichloroethane (DDT), dichlorodiphenyldichloroethylene (DDE) polychlorinated dibenzo-\(p\)-dioxins and -furans (PCDD/F) in all species examined (with a few exceptions).

The chlorinated compounds generally show a somewhat higher concentration in the southern parts of Sweden than in the northern parts.

CB-153 concentration is below the suggested target level of 1.6 µg/g lipid weight in all species and areas, while the target level for CB-118 of 0.024 µg/g lipid weight is exceeded in Lakes Krankesjön, Fysingen, Fräcksjön, Sännen, Tärnan, and Stora Skärsjön and in pike from Lake Bolmen. For DDE the concentration is below the suggested target level of 0.005 µg/g wet weight for all species and areas. ∑HCH is below the suggested target level of 0.026 µg/g wet weight for all species and areas. HCB is below the suggested target level of 0.010 µg/g wet weight for all species and areas. TCDD equivalents (TEQ) is below the suggested target level of 3.5 pg WHO05-TEQ/g wet weight for all species and areas.

**Brominated Flame Retardants**

No general linear trend is observed during the whole monitoring period for the polybrominated diphenyl ethers (PBDEs). However, the concentrations of PBDEs in Lake Bolmen increased from the start of the monitoring period until the late 80s to the mid-90s and appear to have decreased since then. The lower brominated flame retardants (BDE-47, -99 and -100) peaked earlier than the higher (BDE-153 and -154).

In all areas, BDE-47 is above the suggested target level of 0.0085 ng/g wet weight for all species.

The concentration of hexabromocyclododecane (HBCDD) is under LOQ in a majority of the freshwater samples. Hence, it is not possible to draw any conclusions about any trend other than that at least there is not a substantial increase of HBCDD in the biota of the monitored lakes.

**PFASs**

Perfluorononanoic acid (PFNA), perfluorodecanoic acid (PFDA), perfluorotridecanoic acid (PFTrDA) and perfluoroundecanoic acid (PFUnDA) all show significantly increasing concentrations in Arctic char liver from Lake Abiskojaure. PFDA, PFUnDA, perfluorododecanoic acid (PFDoDA), and PFTrDA show upward trends in perch liver from Lake Skärgölen. A downward trend in perfluorooctanesulfonic acid (PFOS) is seen for perch in Lake Skärgölen and for FOSA in perch from Lake Skärgölen during the last nine years.

In all lakes, PFOS is below the suggested EU target level of 9.1 ng/g wet weight for all species.

Information about the lakes sampled within the programme can be seen in Appendix 1.
3 Sampling

3.1 Collected specimens
In general, older specimens show a greater within-year variation compared to younger specimens. To increase the comparability between years, relatively young specimens are collected.

For many species, adults are more mobile than sub-adults. However, the specimens collected need to be of a certain size to allow individual chemical analysis, and thus the size and age of the specimens varies between species and sites (see chapter 4).

To be able to make a selection of individuals of equal size and weight for analysis, about 50 individuals are collected at each site. Only healthy looking specimens with undamaged skin are selected. The collected specimens are placed individually in polyethene plastic bags, deep frozen as soon as possible, and transported to the sample preparation laboratory.

Collected specimens not used in the annual contaminant monitoring programme are stored in the ESB (see (Odsjö, 1993) for further information). A record of these specimens including; biological information, notes about available tissue amounts, together with a precise location in the cold-store are kept and accessible thru a database. These specimens are thus available for retrospective analyses or for control purposes.

Sampling of perch is carried out in the autumn (August–October) outside the spawning season. Char is sampled in the autumn (August–November), which is usually during spawning. Pike is collected in spring (April–May), during or soon after spawning.

Earlier in the programme’s existence, roach were collected from a number of lakes. This was either prior to or during the same time as the collection of perch. Since 2007, collection of roach has ceased. The lakes are shown in Figure 5.1 and 5.2.

3.2 Number of samples and sampling frequency
Previously, 10 specimens were analysed annually from each lake, either individually or as a pooled sample, but from 2011 and onward 12 samples are analysed (individually or as a pool). Historically, individual samples were common, but this has changed. Nowadays, the pooling of samples is done more or less exclusively for organic pollutants. This is mostly due to greater cost-effectiveness, which in turn allows analyses of additional locations and substances (Bignert A. et al., 2014).

Sampling is carried out annually in all time series. The sampling recommendation prescribes a range for age and/or weight of individuals. In a few cases it has not been possible to achieve the required number of individuals within that range. A lower frequency would result in a considerable decrease of statistical and interpretational power. During a period of reduced analytical capacity (2001–2005), several of the collected samples were not analysed but instead stored in the ESB. This situation has now changed, and since 2007 most material is analysed for most substances.

3.3 Sample preparation and registered variables
For each specimen total body weight, total length, body length, sex, age, gonad weight, state of nutrition, liver weight and sample weight are registered (see chapter 4 for descriptions of various age determination methods, depending on species).
The epidermis and subcutaneous fatty tissue are carefully removed. Muscle samples are taken from the middle dorsal muscle layer. The liver is completely removed and weighed. See TemaNord (Nordic Council of Ministers, 1995) for further details about sample preparation.

Fish muscle tissue is analysed for organochlorines (DDTs, PCBs, HCHs, HCB and dioxins), PBDE (Poly Brominated Diphenyl Ethers), HBCDD, and PAH (Polycyclic Aromatic Hydrocarbons). Fish liver is analysed for PFASs (Perfluoroalkyl substances).

In addition to the above analyses, muscle samples are analysed for mercury, and liver samples for lead, cadmium, nickel, chromium, copper, zinc, silver, aluminium, bismuth, tin and arsenic.

### 3.4 Age determination

Age determination in pike is made by reading the age of the cleithrum. In char and perch, the otoliths are used for age determination. After determination, the material is stored at room temperature, and filed using the relevant specimen number. This allows redetermination or a second opinion if there are uncertainties.

### 3.5 Data registration

Data are stored in a flat ASCII file in a hierarchical fashion where each individual specimen represents one level. Each measured value is coded; the codes are defined in a code list. The primary data files are processed through a quality control program. Suspect values are checked and corrected if necessary. Data are retrieved from the primary file into a table format suitable for further import to database or statistical programs.
4 Sample matrices

Of the three species collected, pike has been collected for the longest period. Perch is the most numerous in terms of both the number of collected individuals and the number of lakes.

Table 4.1. Number of individual specimens of various species sampled for analysis of contaminants within the base programme.

<table>
<thead>
<tr>
<th>Species</th>
<th>N of individual specimen</th>
<th>Percent of total %</th>
</tr>
</thead>
<tbody>
<tr>
<td>Pike</td>
<td>1376</td>
<td>20</td>
</tr>
<tr>
<td>Char</td>
<td>806</td>
<td>11</td>
</tr>
<tr>
<td>Perch</td>
<td>4759</td>
<td>69</td>
</tr>
<tr>
<td>Total</td>
<td>6941</td>
<td>100</td>
</tr>
</tbody>
</table>

4.1 Pike (*Esox lucius*)

Male pike become sexually mature between 1–3 years of age; females become sexually mature between 2–5 years of age. Spawning takes place during March–May. Adult pike feed on fish, snakes, frogs and young birds. Pike is a lean fish with an average muscle fat content of 0.56% (geometric mean of all samples).

Pike are collected from two sites: Lake Bolmen in the County of Kronoberg since 1967, and Lake Storvindeln in the County of Västerbotten since 1968 (table 4.2). These two time series are probably the longest series of frozen stored fish in the world. They have been used for retrospective studies of contaminant concentrations for several pollutants.

The specimens from Lake Bolmen are collected during March–May. Specimens from Lake Storvindeln are collected mid-May with few exceptions.

Table 4.2. Number of samples, number of years collected and the arithmetic mean for weight, age and length with 95% confidence intervals for pike analysed at Lake Bolmen and Lake Storvindeln.

<table>
<thead>
<tr>
<th></th>
<th>Lake Bolmen (95% c.i.)</th>
<th>Lake Storvindeln (95% c.i.)</th>
</tr>
</thead>
<tbody>
<tr>
<td>n years</td>
<td>48</td>
<td>47</td>
</tr>
<tr>
<td>Age (years)</td>
<td>5.2 (5.0–5.4)</td>
<td>5.7 (5.5–5.9)</td>
</tr>
<tr>
<td>Length (cm)</td>
<td>54.3 (53.4–55.2)</td>
<td>60.3 (59.4–61.2)</td>
</tr>
<tr>
<td>Weight (g)</td>
<td>1158 (1087–1230)</td>
<td>1383 (1325–1442)</td>
</tr>
</tbody>
</table>

4.2 Arctic char (*Salvelinus alpinus*)

Arctic char become sexually mature between 3–5 years of age. Spawning takes place during August–October. Arctic char muscle tissue is the fattiest of the three species sampled, with an average fat content of about 1.5% (geometric mean of all samples).
Arctic char is collected in autumn from three sites: Lake Abiskojaure (County of Norrbotten) since 1981, Lake Tjulträsk (County of Västerbotten) since 1982, and Lake Stor-Björsjön (County of Jämtland) since 2007 (table 4.3).

Table 4.3. Number of samples, number of years collected and arithmetic mean for weight, age and length with 95% confidence intervals for char analysed at Lakes Abiskojaure, Tjulträsk and Stor-Björsjön.

<table>
<thead>
<tr>
<th></th>
<th>Lake Abiskojaure (95% c.i.)</th>
<th>Lake Tjulträsk (95% c.i.)</th>
<th>Lake Stor-Björsjön (95% c.i.)</th>
</tr>
</thead>
<tbody>
<tr>
<td>n of years</td>
<td>35</td>
<td>34</td>
<td>9</td>
</tr>
<tr>
<td>Age (years)</td>
<td>5.2 (5.0–5.3)</td>
<td>5.2 (5.1–5.3)</td>
<td>5.6 (5.1–5.9)</td>
</tr>
<tr>
<td>Length (cm)</td>
<td>27.4 (27.0–27.9)</td>
<td>26.6 (25.9–27.3)</td>
<td>27.0 (26.1–27.9)</td>
</tr>
<tr>
<td>Weight (g)</td>
<td>226 (212–240)</td>
<td>197 (178–215)</td>
<td>177 (159–196)</td>
</tr>
</tbody>
</table>

4.3 Perch (*Perca fluviatilis*)

Perch is an omnivorous, opportunistic predatory fish. Male perch become sexually mature between 2–4 years of age; females become sexually mature between 3–6 years of age. Spawning takes place during April–June when water temperature is around 7–8°C. Perch muscle tissue is lean and contains approximately 0.4% fat (geometric mean of all samples). Perch is collected from 27 lakes (table 4.4). Sample collection occurs between August–October.
Table 4.4. Number of samples, number of years collected and arithmetic mean for age, length and weight with 95% confidence intervals for perch analysed within the monitoring programme.

<table>
<thead>
<tr>
<th>LAKE</th>
<th>N OF YEARS</th>
<th>AGE (YEARS)</th>
<th>LENGTH (CM) 95% C.I.</th>
<th>WEIGHT (G) 95% C.I.</th>
</tr>
</thead>
<tbody>
<tr>
<td>Allgjuttern</td>
<td>13</td>
<td>4.5 (4.3–4.7)</td>
<td>18.3 (17.9–18.7)</td>
<td>60.0 (55.7–64.2)</td>
</tr>
<tr>
<td>Brännträsket</td>
<td>12</td>
<td>7.5 (7.1–8.0)</td>
<td>18.9 (18.5–19.2)</td>
<td>69.3 (64.3–74.3)</td>
</tr>
<tr>
<td>Bysjön</td>
<td>16</td>
<td>5.4 (5.2–5.7)</td>
<td>17.3 (17.1–17.5)</td>
<td>56.2 (54.0–58.3)</td>
</tr>
<tr>
<td>Bästeträsk</td>
<td>12</td>
<td>4.0 (3.8–4.2)</td>
<td>17.8 (17.4–18.1)</td>
<td>57.0 (53.4–60.5)</td>
</tr>
<tr>
<td>Degervattnet</td>
<td>16</td>
<td>5.9 (5.6–6.2)</td>
<td>17.9 (17.6–18.1)</td>
<td>63.1 (60.1–66.1)</td>
</tr>
<tr>
<td>Fiolen</td>
<td>16</td>
<td>5.0 (4.7–5.3)</td>
<td>18.2 (17.2–18.6)</td>
<td>69.1 (63.2–75.1)</td>
</tr>
<tr>
<td>Fräcksjön</td>
<td>11</td>
<td>5.2 (4.9–5.5)</td>
<td>16.6 (16.4–16.9)</td>
<td>46.8 (44.5–49.2)</td>
</tr>
<tr>
<td>Fysingen</td>
<td>11</td>
<td>4.5 (4.2–4.8)</td>
<td>16.9 (16.6–17.2)</td>
<td>49.9 (46.4–53.3)</td>
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<td>Gipsjön</td>
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<td>6.2 (5.9–6.4)</td>
<td>17.6 (17.3–17.9)</td>
<td>58.9 (55.5–62.2)</td>
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<td>Hjärtsjön</td>
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<td>18.5 (18.2–18.8)</td>
<td>69.8 (65.4–74.3)</td>
</tr>
<tr>
<td>Horsan</td>
<td>11</td>
<td>5.4 (5.1–5.7)</td>
<td>18.2 (17.8–18.5)</td>
<td>60.5 (56.9–64.1)</td>
</tr>
<tr>
<td>Krageholmsjön</td>
<td>16</td>
<td>2.9 (2.6–3.1)</td>
<td>17.0 (16.7–17.4)</td>
<td>63.0 (57.9–68.2)</td>
</tr>
<tr>
<td>Krankesjön</td>
<td>10</td>
<td>3.0 (2.8–3.1)</td>
<td>17.2 (16.9–17.6)</td>
<td>59.9 (55.8–64.0)</td>
</tr>
<tr>
<td>Lilla Öresjön</td>
<td>12</td>
<td>5.6 (5.3–5.9)</td>
<td>18.5 (18.0–19.0)</td>
<td>65.2 (59.3–71.0)</td>
</tr>
<tr>
<td>Limmingsjön</td>
<td>11</td>
<td>5.0 (4.8–5.2)</td>
<td>17.5 (17.2–17.8)</td>
<td>55.2 (52.8–57.6)</td>
</tr>
<tr>
<td>Remmarsjön</td>
<td>16</td>
<td>6.7 (6.4–6.9)</td>
<td>19.3 (18.8–19.9)</td>
<td>84.0 (76.0–92.0)</td>
</tr>
<tr>
<td>Skärgölen</td>
<td>35</td>
<td>4.6 (4.5–4.7)</td>
<td>15.1 (15.0–15.3)</td>
<td>39.2 (37.6–40.8)</td>
</tr>
<tr>
<td>Spjutsjön</td>
<td>9</td>
<td>3.6 (3.4–3.8)</td>
<td>18.1 (17.8–18.4)</td>
<td>61.2 (57.5–64.9)</td>
</tr>
<tr>
<td>Stora Envättern</td>
<td>16</td>
<td>5.9 (5.7–6.2)</td>
<td>17.1 (16.8–17.3)</td>
<td>52.8 (50.1–55.5)</td>
</tr>
<tr>
<td>Stensjön</td>
<td>19</td>
<td>7.0 (6.7–7.2)</td>
<td>18.2 (18.0–18.4)</td>
<td>60.7 (58.7–62.6)</td>
</tr>
<tr>
<td>Stora Skärsvet</td>
<td>15</td>
<td>6.4 (6.1–6.6)</td>
<td>16.8 (16.5–17.1)</td>
<td>52.3 (48.9–55.7)</td>
</tr>
<tr>
<td>Stor-Backsjön</td>
<td>12</td>
<td>6.1 (5.9–6.4)</td>
<td>18.0 (17.8–18.3)</td>
<td>59.8 (57.2–62.4)</td>
</tr>
<tr>
<td>Svartsjön</td>
<td>10</td>
<td>6.0 (5.4–6.6)</td>
<td>15.6 (14.6–16.6)</td>
<td>43.9 (32.0–55.9)</td>
</tr>
<tr>
<td>Sännen</td>
<td>12</td>
<td>5.5 (5.2–5.8)</td>
<td>17.0 (16.7–17.3)</td>
<td>47.7 (43.8–51.6)</td>
</tr>
<tr>
<td>Tärnan</td>
<td>16</td>
<td>6.1 (5.7–6.5)</td>
<td>17.4 (17.0–17.8)</td>
<td>55.3 (50.8–59.9)</td>
</tr>
<tr>
<td>Algsvet</td>
<td>11</td>
<td>5.8 (5.4–6.2)</td>
<td>17.0 (16.7–17.3)</td>
<td>50.5 (47.7–53.3)</td>
</tr>
<tr>
<td>Övre Skärsvet</td>
<td>16</td>
<td>6.9 (6.5–7.3)</td>
<td>17.9 (17.6–18.1)</td>
<td>58.9 (56.4–61.4)</td>
</tr>
</tbody>
</table>
5 Sampling sites

Location and names of the sample sites are shown in figure 5.1. The sampling sites are selected following the below criteria:

- lakes must not be influenced by local contamination;
- land use in the areas surrounding the lake should be well investigated and intensive rural areas avoided;
- lakes should preferably be placed high in the drainage system;
- the influence of liming activities should be avoided;
- lakes should have some protection against future exploitation;
- oligotrophic lakes are preferred;
- To facilitate regional comparisons, the selected lakes should preferably be as similar as possible concerning factors that could influence the concentration of various contaminants in the analysed biological tissues.

Generally, fish from eutrophic lakes show a slower response to changes in the amount of discharge in the lake compared to oligotrophic lakes, thus explaining the preference for oligotrophic lakes for these monitoring activities.

Sample collection occurs in 32 lakes distributed from north to south of Sweden (Fig. 5.1). Lakes where samples were collected earlier, but are no longer sampled, are shown in figure 5.2. The discontinued collection is mostly due to intentional changes e.g., liming. Samples from these locations are, however, still available in the ESB and could be used for comparisons and retrospective analyses. More information about the lakes currently sampled in the programme can be found in Appendix 1.
**Figure 5.1.** Map showing lake location, including species and year, within the Swedish National Monitoring Programme.

* Roach has been collected before and/or at the same time as perch.
**Figure 5.2.** Location of lakes where sampling has been discontinued, including species sampled and years. In Lake Åmten, perch, roach and pike were collected during the stated years.

<table>
<thead>
<tr>
<th>NR</th>
<th>Sampling site</th>
<th>Species</th>
<th>Year</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>Stormyrån</td>
<td>Perch</td>
<td>1999-1999</td>
</tr>
<tr>
<td>2</td>
<td>Bysjön, Ockelbo</td>
<td>Perch</td>
<td>2005-2006</td>
</tr>
<tr>
<td>3</td>
<td>Ämte</td>
<td>P.R.P</td>
<td>1982-2000</td>
</tr>
<tr>
<td>4</td>
<td>Bylysjön</td>
<td>Perch</td>
<td>1982-96, 2006</td>
</tr>
<tr>
<td>5</td>
<td>Norra Yngem</td>
<td>Perch</td>
<td>2005-2006</td>
</tr>
<tr>
<td>6</td>
<td>Rotehagstjärn</td>
<td>Perch</td>
<td>1997-99</td>
</tr>
<tr>
<td>7</td>
<td>Kvarnsjön</td>
<td>Perch</td>
<td>1981-96</td>
</tr>
<tr>
<td>8</td>
<td>Stora Sjö</td>
<td>Perch</td>
<td>1984-96</td>
</tr>
<tr>
<td>9</td>
<td>Stora Pipsjön</td>
<td>Perch</td>
<td>1987-96</td>
</tr>
<tr>
<td>10</td>
<td>Fåglasjön</td>
<td>Perch</td>
<td>2005-2006</td>
</tr>
<tr>
<td>11</td>
<td>Krankasjön</td>
<td>Roach</td>
<td>1999-2007</td>
</tr>
</tbody>
</table>
6 Analytical methods

6.1 Organochlorines and brominated flame retardants

The analyses of organochlorines and brominated flame retardants are carried out at the Department of Environmental Science and Analytical Chemistry (ACES) at Stockholm University. Specific analytical methods applied are described in the respective chapters where applicable. Before 1988, organochlorines were analysed by a packed column gas chromatography (GC). During 1988, analysis on a capillary column was introduced, allowing analysis of individual congeners (Eriksson U. et al., 1994). The extraction method originates from the method described by Jensen et al. (Jensen S. et al., 1983) where wet tissues are extracted with a mixture of polar and non-polar solvents. The organochlorines are analysed on a gas chromatograph (GC) equipped with a μ-electron capture detector (Eriksson U. et al., 1994). The BFRs are analysed by a GC connected to a mass spectrometer operating in electron capture negative ionization mode (NICI) (Sellström et al., 1998).

6.1.1 Quality assurance

Quality control for organochlorines has continuously improved over the last 20 years, resulting in accreditation in 1999. Assessment is performed once a year by the accreditation body SWEDAC. The laboratory is fulfilling the obligations in SS-EN IEC 17025:2005. The accreditation is valid for CB-28, 52, 101, 118, 153, 138, 180, HCB, p,p'-DDE, p,p'-DDD, p,p'-DDT and α-, β- and γ-HCH in biological tissues. So far the BFRs are not accredited but the analysis of BDE-47, 99,100, 153, 154 and HBCDD are in many ways performed with the same quality aspects as the organochlorines.

The Quality Assurance program is based on the Quality Manual, standard operation procedures (SOPs) and supplements. The annual audit includes a review of the SOPs, reference materials, proficiency testing, filing system, qualifications of the staff, up-to-date record of the training of the staff (to be able to perform their assigned tasks), accredited methods and audit of the quality program.

6.1.2 Standards

The original of all standards are well documented with known purity and certified concentration with uncertainty for the solutions.

6.1.3 Selectivity

To have the possibility to control impurities in solvents, equipments and glassware, one blank sample is extracted together with each batch of environmental samples. Coelution of PCB congeners and pesticides in GC analysis is dependent upon instrumental conditions such as column type, length, internal diameter, film thickness and oven temperature. To minimize possible coelutions, two 60 m columns are used in parallel, the commonly used 5 % phenyl-methylsilicone phase and the more polar 14 % cyanopropylphenyl-methylsilicone phase. The only remaining known coelution is for CB-138, which coelutes with CB-163 (Larsen and Riego, 1990). Hence, CB-138 and CB-163 is reported as CB138+163. PBDE and HBCDD are analysed on a 30 m DB-5 MS column, monitoring m/z 79 and 81.
When introducing a new matrix one of the samples is re-extracted with a mixture of more polar solvents for control of no remaining contaminants in the matrix residual.

Samples from new matrixes / new sampling locations are also examined for suitable internal standards.

6.1.4 Reference Material

Three laboratory reference materials (LRM) are used as extraction controls, chosen with respect to their lipid content and level of contaminants. The controls consist of herring and pike respectively salmon muscle, homogenised in a household mixer and stored in aliquots in airtight bags of aluminium laminate at -80°C. At every extraction event one extraction control is extracted as well.

6.1.5 Proficiency testing

Concerning PCBs and pesticides, the laboratory has participated in the periodic QUASIMEME proficiency testing since 1993, with two rounds every year, each one containing two samples. Around 95% of all reported values have been satisfactory according to QUASIMEME, meaning they have been within +/- 2 standard deviations of the assigned value. In 2000, the laboratory participated in the first interlaboratory study ever performed for PBDEs and HBCDD, contaminants that since 2001 are incorporated in the QUASIMEME proficiency testing scheme. Around 85% of the values the laboratory has produced during the years have been satisfactory according to QUASIMEME.

6.1.6 Quantification limits and uncertainty in the measurements

Calculation of the uncertainty in the measurement is based on the Nordtest Report TR 537 “Handbook for calculation of measurement uncertainty in environmental laboratories”, where the within-laboratory reproducibility is combined with estimate of the method and laboratory bias. The within-laboratory reproducibility is calculated from LRM from more than 8000 PCB and pesticide values during a period of nearly 20 years and around 2000 BDE and HBCDD values during nearly 15 years. The bias is estimated from proficiency testing of more than 8 samples during at least 4 years. The bias for PBDE is used also for HBCDD since no reliable proficiency testing (or certified reference material) exists today. Finally, the expanded uncertainty is calculated, using a coverage factor of 2 to reach approximately 95% confidence level (table 6.3). The reproducibility for the PCBs and pesticides follows the theory stated by Horwitz where the relative standard deviation increase when the concentration level decrease (Horwitz and Albert, 2006). The reproducibility for the PBDEs and HBCDD follows a function where the relative standard deviations increase first at the very lowest concentration.

<table>
<thead>
<tr>
<th>Table 6.3. Expanded uncertainty (%) at different concentrations</th>
</tr>
</thead>
<tbody>
<tr>
<td>ng/g lw</td>
</tr>
<tr>
<td>2–50</td>
</tr>
<tr>
<td>&gt; 25</td>
</tr>
<tr>
<td>&gt; 25</td>
</tr>
<tr>
<td>&gt; 25</td>
</tr>
</tbody>
</table>

The quantification limit is estimated to approximately 2 ng/g fat weight for all analysed PCBs, α, β, γ–HCH, HCB, pp-DDE and pp-DDD and 4 ng/g fat weight for pp-DDT. For all
analysed PBDEs the quantification limit is estimated to approximately 0.2 ng/g fat weight and for HBCDD 2 ng/g.

6.2 Dioxins, dibenzofurans and dioxin-like PCBs
The analyses of dioxins and dioxin-like PCBs are carried out at the Department of Chemistry, Umeå University. The extraction method is described by Wiberg et al. (Wiberg et al., 1998), the clean-up method by Danielsson et al. (Danielsson et al., 2005), and the instrumental analysis (GC-HRMS) by Liljelind et al. (Liljelind et al., 2003). The laboratory participates in the annual FOOD intercalibration rounds, including laboratory reference material (salmon tissue) with each set of samples.

6.3 Perfluoroalkyl substances
The analyses of perfluoroalkyl substances are carried out at the Department of Environmental Science and Analytical Chemistry (ACES; formerly the Department of Applied Environmental Science), Stockholm University.

6.3.1 Sample preparation and instrumental analysis
A sample aliquot of approximately 1.0 g homogenized tissue in a polypropylene (PP) centrifuge tube was spiked with 10 ng each of a suite of mass-labelled internal standards (\(^{18}\)O- or \(^{13}\)C-labelled perfluoroalkyl sulfonates and carboxylic acids). The samples were extracted twice with 5 mL of acetonitrile in an ultrasonic bath. Following centrifugation, the supernatant extract was removed and the combined acetonitrile phases were concentrated to 1 mL under a stream of nitrogen. The concentrated extract underwent dispersive clean-up on graphitised carbon and acetic acid. A volume of 0.5 mL of the cleaned-up extract was added to 0.5 mL of aqueous ammonium acetate and volumetric standards M8PFOA and M8PFOS were added. Precipitation during overnight freezing occurred and the extract was centrifuged before the clear supernatant was transferred to an autoinjector vial for instrumental analysis. Aliquots of the final extracts were injected automatically on an ultra performance liquid chromatography (UPLC) system (Acquity, Waters) coupled to a tandem mass spectrometer (MS-MS; Xevo TQS, Waters). Chromatographic separation of target analytes was achieved on a BEH C18 UPLC column (1.7 µm particles, 50 × 2.1 mm, Waters) with a binary gradient of ammonium acetate buffered methanol and water. The mass spectrometer was operated in negative electrospray ionisation, multiple reaction monitoring (MRM) mode. Quantification was performed by isotope dilution or an internal standard approach depending on the target analyte.

6.3.2 Quality control
The extraction method employed in the present study (with the exception of the concentration step) has previously been validated for biological matrices and showed excellent analyte recoveries ranging between 90 and 110% for PFCAs from C6 to C14 (Powley et al., 2005). Including extract concentrations, we determined recoveries between 70 and 90% for C6 to C10-PFCAs and 65–70% for C11–C15 PFCAs. Extraction efficiencies for perfluorosulfonates (PFSAs), including perfluorooctane sulfonamide (FOSA), were determined to 70–95%. Method quantification limits (MQLs) for all analytes were determined on the basis of blank extraction experiments and ranged between 0.05 and 0.5 ng/g wet weight.
for the different compounds. An in-house fish tissue sample which has been analyzed for several years was included as control sample along with all sample batches. The obtained concentrations were in good agreement with the mean concentrations from the ILC study for all seven compounds quantified in the ILC.

6.4 Trace metals

The analyses of trace metals are carried out at the Analytical Environmental Chemistry Unit at the Department of Applied Environmental Science (ITM), University of Stockholm.

6.4.1 Sample preparation and instrumental analysis

Analytical methods for metals in liver are performed according to the Swedish standards SS-EN 13805 (Foodstuffs – Determination of trace elements – Pressure digestion) and SS-EN ISO 17294-2 (Water quality – Application of inductively coupled plasma mass spectrometry (ICP-MS) – Part 2: Determination of 62 elements), and for mercury according to the US EPA Method 7473 (mercury in solids and solutions by thermal decomposition, amalgamation and atomic absorption spectrophotometry).

6.4.2 Quality control

The laboratory participates in the periodic QUASIMEME intercalibration rounds.

6.4.3 Reference Material

CRMs (certified reference material) used for mercury are:
DORM-2 and DORM-3 (dogfish muscle)
For all other metals, CRMs used are:
DOLT-3 (dogfish liver)
NIST 1566 (oyster tissue)
TORT-2 (lobster hepatopancreas)
7 Statistical treatment and graphical presentation

7.1 Trend detection

One of the main objectives of the monitoring programme is to detect trends. Trend detection is carried out in three steps.

7.1.1 Log-linear regression analyses

Log-linear regression analyses are performed for the entire investigated time period and also for the most recent ten years for longer time series.

The slope of the line describes the yearly percentage change. A slope of 5% implies that the concentration is halved in 14 years, whereas 10% corresponds to a similar reduction in 7 years, and 2% in 35 years (table 7.1).

<table>
<thead>
<tr>
<th></th>
<th>1%</th>
<th>2%</th>
<th>3%</th>
<th>4%</th>
<th>5%</th>
<th>7%</th>
<th>10%</th>
<th>12%</th>
<th>15%</th>
<th>20%</th>
</tr>
</thead>
<tbody>
<tr>
<td>Increase</td>
<td>70</td>
<td>35</td>
<td>24</td>
<td>18</td>
<td>14</td>
<td>10</td>
<td>7</td>
<td>6</td>
<td>5</td>
<td>4</td>
</tr>
<tr>
<td>Decrease</td>
<td>69</td>
<td>35</td>
<td>23</td>
<td>17</td>
<td>14</td>
<td>10</td>
<td>7</td>
<td>6</td>
<td>4</td>
<td>3</td>
</tr>
</tbody>
</table>

7.1.2 Non-parametric trend test

The regression analysis presumes, among other things, that the regression line gives a good description of the trend. The leverage affect of points at the end of the line is a well-known fact. An exaggerated slope, caused 'by chance' by a single or a few points at the end of the line increases the risk of a false significant result when no real trend exist. A non-parametric alternative to the regression analysis is the Mann-Kendall trend test (Gilbert, 1987, Helsel D.R. and Hirsch R.M., 1992b, International Council for the Exploration of the Sea, 1995). This test generally has lower power than the regression analysis, and does not take into account differences in magnitude of the concentrations – it only counts the number of consecutive years where the concentration increases or decreases compared with the year before. If the regression analysis yields a significant result but the Mann-Kendall test does not, the explanation could be either that the latter test has lower power, or that the influence of endpoints in the time series has become unaccountably large on the slope. Hence, the eighth line reports Kendall's ‘τ’, and the corresponding p-value. The Kendall's ‘τ’ ranges from 0 to 1 like the traditional correlation coefficient ‘r’ but will generally be lower. ‘Strong’ linear correlations of 0.9 or above, corresponds to τ-values of about 0.7 or above (Helsel D.R. and Hirsch R.M., 1992b). This test was recommended by the Swedish EPA for use in water quality monitoring programmes using annual samples in an evaluation comparing several other trend tests (Loftis J.C. et al., 1989).

7.1.3 Non-linear trend components

An alternative to the regression line used to describe development over time is a type of smoothed line. The smoother applied here is a simple 3-point running mean smoother fitted to the annual geometric mean values. In cases where the regression line is a poor fit, the
smoothed line may be more appropriate. The significance of this line is tested using an ANOVA, where the variance explained by the smoother and the regression line is compared with the total variance. This procedure is used in assessments at ICES and is described by Nicholson et al. (Nicholson et al., 1998).

7.2 Outliers and values below the detection limit

Observations further from the regression line than what is expected from the residual variance around the line are subject to special concern. These deviations may be caused by an atypical occurrence of something in the physical environment, a changed pollution load, or errors in the sampling or analytical procedure. The procedure to detect suspected outliers in this context is described by Hoaglin and Welsch (Hoaglin and Welsch, 1978). It makes use of the leverage coefficients and the standardised residuals. The standardised residuals are tested against a $t_{0.05}$ distribution with $n-2$ degrees of freedom. When calculating the $i$th standardised residual, the current observation is left out, implying that the $i$th observation does not influence the slope or the variance around the regression line. The suspected outliers are merely indicated in the figures and are included in the statistical calculations, except in a very few cases, which are pointed out in the figures.

Values reported that are below the quantification limit are substituted using the reported LOQ (or in the case that this information is missing the minimum value for the current year) divided by the square root of 2.

7.3 Plot Legends

Analytical results from each of the investigated elements are displayed graphically. A selection of sites and species are presented as figures; no time series is shorter than four years.

Each figure displays the geometric mean concentration of each year (circles) together with the individual analyses (small dots) and the 95% confidence intervals of the geometric means. The overall geometric mean value for the time series is depicted as a thin horizontal line.

The trend for the whole time period is presented by a regression line (plotted if $p < 0.10$, two-sided regression analysis); $p < 0.05$ is presented by a red line and $0.05 < p < 0.10$ is presented by a dashed blue line. The trend for the last ten years is plotted if $p < 0.2$ and $p < 0.05$ is presented by a red line and $0.05 < p < 0.2$ is presented by a dashed light blue line. Ten years is often a too short period to statistically detect a trend unless it is of considerable magnitude. Nevertheless, the ten year regression line will indicate a possible change in the direction of a trend. Furthermore, the residual variance around the line compared to the residual variance for the entire period will indicate if the sensitivity has increased as a result of e.g., improved sampling technique or that problems in the chemical analysis have disappeared.

A smoother is applied to test for non-linear trend components (see section 7.1.3). The smoothed line is plotted if $p < 0.10$ and $p < 0.05$ is presented by a red line and $0.05 < p < 0.10$ is presented by a dashed blue line. A broken line segment indicates a gap in the time series with a missing year.

The log-linear regression lines fitted through the geometric mean concentrations follow smooth exponential functions.
A cross inside a circle indicates a suspected outlier (see section 7.3). Suspected outliers are indicated in the figures and are included in the statistical analyses except in a few cases, as pointed out in the figures.

Each figure has a header with species name and sampling locality. Below the header, the results from several statistical analyses are reported:

- **Tv** = \ldots \text{lp}\% or \text{dp}\% = \ldots \text{Tv} is the target level (see Chapter 10) calculated on a lipid weight base (\text{lp}\%) or on dry weight base (\text{dp}\%), original target value was given on a wet weight basis.

- \text{n(tot)} = \text{first line reports the total number of analyses included together with the number of years (n(yrs)}) ;

- \text{m} = \text{overall geometric mean value together with its 95% confidence interval is reported on the second line of the plot (N.B. d.f. = number of years -1)};

- \text{slope} = \text{reports the slope, expressed as the yearly percentage change together with its 95% confidence interval};

- \text{CV} = \text{reports the coefficient of variation around the regression line, as a measure of between-year variation, together with the lowest detectable change in the current time series with a power of 80\%, one-sided test, } \alpha=0.05. \text{ The last figure on this line is the estimated number of years required to detect an annual change of 10\% with a power of 80\%, one-sided test, } \alpha=0.05.

- \text{power} = \text{reports the power to detect a log-linear trend in the time series (Nicholson and Fryer, 1992). The first number represents the power to detect an annual change of 5\% with the number of years in the current time series. The second number is the power estimated as if the slope where 5\% a year and the number of years were ten. The third number is the lowest detectable change (given in percent per year) for a ten year period with the current between year variation at a power of 80\%. The results of the power analyses from the various time series are summarised in chapter 8};

- \text{r}^2 = \text{reports the coefficient of determination (r}^2\text{) together with a p-value for a two-sided test (H}_0: \text{slope} = 0 \text{) i.e. a significant value is interpreted as a true change, provided that the assumptions of the regression analysis are fulfilled};

- \text{y(11)} = \text{reports the concentration estimated from the regression line for the last year together with a 95\% confidence interval e.g., } y(06)=2.55(2.17, 3.01) \text{ is the estimated concentration of year 2006 where the residual variance around the regression line is used to calculate the confidence interval. Provided that the regression line is relevant to describe the trend, the residual variance might be more appropriate than the within-year variance in this respect};

- \text{tao} = \text{reports Kendall’s ‘r’, and the corresponding p-value};

- \text{sd(sm)} = \text{reports the coefficient of variation around the smoothed line. The significance of this line can be tested by means of an ANOVA (see section 7.1.3). The p-value is reported for this test. A significant result will indicate a non-linear trend component. After the p-value, the minimum trend (percentage per year) likely to be detected at a power of 80\% during a period of 10 years, should a log-linear trend occur, is shown. This estimate is compensated for the loss of degrees of freedom, considering the smoother}.
Below these nine lines are additional lines with information concerning the regression of the last ten years.

In a few cases where an extreme outlying observation may compromise the confidence in the regression line, the ordinary regression line is replaced by the ‘Kendall-Theil Robust line’ (Helsel D.R. and Hirsch R.M., 1992a). In these cases only the ‘Theil’-slope and Kendall’s ‘τ’ are reported.

7.4 Legend for the three dimensional maps

The height of the bars represents the arithmetic mean for the last three years, or less if results are not available. The bars are split into three sections.

Three dimensional maps with target levels are only presented if the target level lies within the concentration range. The green section of the bars denotes arithmetic mean concentrations (for the last three years) below the suggested target level and the red sections denote arithmetic mean concentrations (for the last three years) above the suggested target level.
8 The power of the programme

Before starting to interpret the results from the statistical analyses of the time series, it is essential to know with what power temporal changes can be detected (i.e., the chance to reveal true trends with the investigated matrices). It is crucial to know whether a negative result of a trend test indicates a stable situation, or if the monitoring programme is too poor to detect even serious changes in the contaminant load in the environment. One approach to this problem is to estimate the power of the time series based on the ‘random’ between-year variation. Alternatively, the lowest detectable trend could be estimated at a fixed power to represent the sensitiveness of the time series.

The first task would thus be to estimate the ‘random’ between-year variation. In the results presented below, this variation is calculated using the residual distance from a log-linear regression line. In many cases the log-linear line, fitted to the current observations, seems to be an acceptable ‘neutral’ representation of the true development of the time series. In cases where a significant ‘non-linear’ trend has been detected (see chapter 7) the regression line may not serve this purpose; hence the sensitivity and/or power results based on such time series are marked with an asterix in the following tables. These results are also excluded from estimations of median performances.

Another problem is that a single outlier could ruin the estimation of the between-year variation. In the present results suspected outliers are included, which means that the power and sensitiveness might be underestimated.

Results from the longest timeseries for most of the contaminants, Lake Abiskojaure (Arctic char), Lake Bolmen and Lake Storvindeln (pike) and Lake Skärgölen (perch), are presented in Tabel 8.1, 8.2, 8.3 and 8.4.

Table 8.1 shows the number of years that various contaminants have been analysed and detected from the monitored sites.
Table 8.1. Number of years that various contaminants have been analysed and detected.

<table>
<thead>
<tr>
<th>Contaminant</th>
<th>Lake Abiskojaure</th>
<th>Lake Storvindeln</th>
<th>Lake Bolmen</th>
<th>Lake Skärgölen</th>
</tr>
</thead>
<tbody>
<tr>
<td>Mercury (Hg)</td>
<td>34</td>
<td>43</td>
<td>35</td>
<td>29</td>
</tr>
<tr>
<td>Lead (Pb)</td>
<td>34</td>
<td>44</td>
<td>18</td>
<td>15</td>
</tr>
<tr>
<td>Cadmium (Cd)</td>
<td>34</td>
<td>444</td>
<td>18</td>
<td>15</td>
</tr>
<tr>
<td>Nickel (Ni)</td>
<td>34</td>
<td>44</td>
<td>18</td>
<td>15</td>
</tr>
<tr>
<td>Chromium (Cr)</td>
<td>34</td>
<td>44</td>
<td>18</td>
<td>15</td>
</tr>
<tr>
<td>Copper (Cu)</td>
<td>34</td>
<td>44</td>
<td>18</td>
<td>15</td>
</tr>
<tr>
<td>Zink (Zn)</td>
<td>34</td>
<td>44</td>
<td>18</td>
<td>15</td>
</tr>
<tr>
<td>Arsenic (As)</td>
<td>16</td>
<td>15</td>
<td>13</td>
<td>14</td>
</tr>
<tr>
<td>Silver (Ag)</td>
<td>13</td>
<td>12</td>
<td>12</td>
<td>14</td>
</tr>
<tr>
<td>Aluminium (Al)</td>
<td>16</td>
<td>15</td>
<td>13</td>
<td>14</td>
</tr>
<tr>
<td>Bismuth (Bi)</td>
<td>8</td>
<td>7</td>
<td>7</td>
<td>9</td>
</tr>
<tr>
<td>Tin (Sn)</td>
<td>12</td>
<td>11</td>
<td>11</td>
<td>13</td>
</tr>
<tr>
<td>CB-153</td>
<td>26</td>
<td>25</td>
<td>24</td>
<td>10</td>
</tr>
<tr>
<td>DDE</td>
<td>30</td>
<td>44</td>
<td>45</td>
<td>16</td>
</tr>
<tr>
<td>γ-HCH</td>
<td>26</td>
<td>18</td>
<td>24</td>
<td>10</td>
</tr>
<tr>
<td>HCB</td>
<td>26</td>
<td>25</td>
<td>24</td>
<td>10</td>
</tr>
<tr>
<td>TCDD-eqv (PCDD/PCDF)</td>
<td>-</td>
<td>15</td>
<td>14</td>
<td>20</td>
</tr>
<tr>
<td>PFOS</td>
<td>22</td>
<td>-</td>
<td>-</td>
<td>21</td>
</tr>
<tr>
<td>BDE-47</td>
<td>25</td>
<td>-</td>
<td>40</td>
<td>-</td>
</tr>
</tbody>
</table>

Table 8.2 shows the number of years required to detect an annual change of 10% with a power of 80%. The power is to a great extent dependent of the length of the time series. The possibility to statistically verify an annual change of 10% at a power of 80% generally requires 10–20 years for organic substances.

Table 8.2. The number of years required to detect an annual change of 10% with a power of 80%.

<table>
<thead>
<tr>
<th>Contaminant</th>
<th>Lake Abiskojaure</th>
<th>Lake Storvindeln</th>
<th>Lake Bolmen</th>
<th>Lake Skärgölen</th>
</tr>
</thead>
<tbody>
<tr>
<td>Mercury (Hg)</td>
<td>10</td>
<td>11</td>
<td>9</td>
<td>12</td>
</tr>
<tr>
<td>Lead (Pb)</td>
<td>15</td>
<td>14</td>
<td>16</td>
<td>12</td>
</tr>
<tr>
<td>Cadmium (Cd)</td>
<td>17</td>
<td>12</td>
<td>15</td>
<td>14</td>
</tr>
<tr>
<td>Nickel (Ni)</td>
<td>15</td>
<td>13</td>
<td>15</td>
<td>15</td>
</tr>
<tr>
<td>Chromium (Cr)</td>
<td>17</td>
<td>18</td>
<td>22</td>
<td>24</td>
</tr>
<tr>
<td>Copper (Cu)</td>
<td>14</td>
<td>11</td>
<td>11</td>
<td>10</td>
</tr>
<tr>
<td>Zink (Zn)</td>
<td>19</td>
<td>9</td>
<td>14</td>
<td>7</td>
</tr>
<tr>
<td>Arsenic (As)</td>
<td>13</td>
<td>14</td>
<td>10</td>
<td>18</td>
</tr>
<tr>
<td>Silver (Ag)</td>
<td>14</td>
<td>11</td>
<td>14</td>
<td>11</td>
</tr>
<tr>
<td>Aluminium (Al)</td>
<td>14</td>
<td>15</td>
<td>10</td>
<td>12</td>
</tr>
<tr>
<td>Bismuth (Bi)</td>
<td>7</td>
<td>13</td>
<td>11</td>
<td>14</td>
</tr>
<tr>
<td>Tin (Sn)</td>
<td>15</td>
<td>17</td>
<td>15</td>
<td>14</td>
</tr>
<tr>
<td>CB-153</td>
<td>12</td>
<td>10</td>
<td>12</td>
<td>8</td>
</tr>
<tr>
<td>DDE</td>
<td>14</td>
<td>12</td>
<td>14</td>
<td>16</td>
</tr>
<tr>
<td>γ-HCH</td>
<td>18</td>
<td>12</td>
<td>10</td>
<td>7</td>
</tr>
<tr>
<td>HCB</td>
<td>10</td>
<td>11</td>
<td>13</td>
<td>5</td>
</tr>
<tr>
<td>TCDD-eqv (PCDD/PCDF)</td>
<td>-</td>
<td>13</td>
<td>12</td>
<td>13</td>
</tr>
<tr>
<td>PFOS</td>
<td>15</td>
<td>-</td>
<td>-</td>
<td>15</td>
</tr>
<tr>
<td>BDE-47</td>
<td>26</td>
<td>-</td>
<td>16</td>
<td>-</td>
</tr>
</tbody>
</table>
The lowest trend that is possible to detect within a 10 year period with a power of 80% is presented for the entire time series (table 8.3).

Table 8.3. The lowest trend possible to detect (in %) within a 10 year period with a power of 80% for the entire time series.

<table>
<thead>
<tr>
<th>Contaminant</th>
<th>Lake Abiskojaure</th>
<th>Lake Storvindeln</th>
<th>Lake Bolmen</th>
<th>Lake Skärgölen</th>
</tr>
</thead>
<tbody>
<tr>
<td>Mercury (Hg)</td>
<td>9.6</td>
<td>11</td>
<td>8.3</td>
<td>15</td>
</tr>
<tr>
<td>Lead (Pb)</td>
<td>21</td>
<td>20</td>
<td>23</td>
<td>14</td>
</tr>
<tr>
<td>Cadmium (Cd)</td>
<td>27</td>
<td>13</td>
<td>21</td>
<td>18</td>
</tr>
<tr>
<td>Nickel (Ni)</td>
<td>22</td>
<td>16</td>
<td>23</td>
<td>22</td>
</tr>
<tr>
<td>Chromium (Cr)</td>
<td>29</td>
<td>31</td>
<td>47</td>
<td>64</td>
</tr>
<tr>
<td>Copper (Cu)</td>
<td>18</td>
<td>11</td>
<td>13</td>
<td>9.5</td>
</tr>
<tr>
<td>Zink (Zn)</td>
<td>8.4</td>
<td>8.5</td>
<td>19</td>
<td>3.7</td>
</tr>
<tr>
<td>Arsenic (As)</td>
<td>15</td>
<td>19</td>
<td>9.5</td>
<td>32</td>
</tr>
<tr>
<td>Silver (Ag)</td>
<td>20</td>
<td>11</td>
<td>20</td>
<td>11</td>
</tr>
<tr>
<td>Aluminium (Al)</td>
<td>19</td>
<td>22</td>
<td>10</td>
<td>15</td>
</tr>
<tr>
<td>Bismuth (Bi)</td>
<td>4.3</td>
<td>16</td>
<td>11</td>
<td>19</td>
</tr>
<tr>
<td>Tin (Sn)</td>
<td>21</td>
<td>27</td>
<td>23</td>
<td>19</td>
</tr>
<tr>
<td>CB-153</td>
<td>13</td>
<td>8.6</td>
<td>14</td>
<td>6.7</td>
</tr>
<tr>
<td>DDE</td>
<td>18</td>
<td>14</td>
<td>20</td>
<td>25</td>
</tr>
<tr>
<td>γ-HCH</td>
<td>32</td>
<td>15</td>
<td>10</td>
<td>4.4</td>
</tr>
<tr>
<td>HCB</td>
<td>10</td>
<td>12</td>
<td>16</td>
<td>2.1</td>
</tr>
<tr>
<td>TCDD-equiv (PCDD/PCDF)</td>
<td>-</td>
<td>13</td>
<td>14</td>
<td>17</td>
</tr>
<tr>
<td>PFOS</td>
<td>22</td>
<td>-</td>
<td>-</td>
<td>23</td>
</tr>
<tr>
<td>BDE-47</td>
<td>71</td>
<td>-</td>
<td>24</td>
<td>-</td>
</tr>
</tbody>
</table>

The power to detect an annual change of 10% for the monitoring period i.e., the length of the time series varies depending on site and investigated contaminant (table 8.4). For the longest time series, the estimated power is close to 100% in most cases.
Table 8.4. Power to detect an annual change of 10% for the entire monitoring period. The length of the time series varies depending on site and investigated contaminant. In cases where considerable increased power has been achieved during the most recent ten years period, this value has been used.

<table>
<thead>
<tr>
<th>Contaminant</th>
<th>Lake Abiskojaure</th>
<th>Lake Storvindeln</th>
<th>Lake Bolmen</th>
<th>Lake Skärgölen</th>
</tr>
</thead>
<tbody>
<tr>
<td>Mercury (Hg)</td>
<td>1.0</td>
<td>1.0</td>
<td>1.0</td>
<td>1.0</td>
</tr>
<tr>
<td>Lead (Pb)</td>
<td>1.0</td>
<td>1.0</td>
<td>0.97</td>
<td>1.0</td>
</tr>
<tr>
<td>Cadmium (Cd)</td>
<td>1.0</td>
<td>1.0</td>
<td>0.99</td>
<td>0.94</td>
</tr>
<tr>
<td>Nickel (Ni)</td>
<td>1.0</td>
<td>1.0</td>
<td>0.98</td>
<td>0.85</td>
</tr>
<tr>
<td>Chromium (Cr)</td>
<td>1.0</td>
<td>1.0</td>
<td>0.55</td>
<td>0.20</td>
</tr>
<tr>
<td>Copper (Cu)</td>
<td>1.0</td>
<td>1.0</td>
<td>1.0</td>
<td>1.0</td>
</tr>
<tr>
<td>Zink (Zn)</td>
<td>1.0</td>
<td>1.0</td>
<td>0.96</td>
<td>1.0</td>
</tr>
<tr>
<td>Arsenic (As)</td>
<td>1.0</td>
<td>0.93</td>
<td>1.0</td>
<td>0.43</td>
</tr>
<tr>
<td>Silver (Ag)</td>
<td>0.69</td>
<td>0.98</td>
<td>0.57</td>
<td>1.0</td>
</tr>
<tr>
<td>Aluminium (Al)</td>
<td>0.97</td>
<td>0.85</td>
<td>1.0</td>
<td>0.97</td>
</tr>
<tr>
<td>Bismuth (Bi)</td>
<td>0.99</td>
<td>0.13</td>
<td>0.25</td>
<td>0.24</td>
</tr>
<tr>
<td>Tin (Sn)</td>
<td>0.53</td>
<td>0.26</td>
<td>0.35</td>
<td>0.72</td>
</tr>
<tr>
<td>CB-153</td>
<td>1.0</td>
<td>1.0</td>
<td>1.0</td>
<td>0.99</td>
</tr>
<tr>
<td>DDE</td>
<td>1.0</td>
<td>1.0</td>
<td>1.0</td>
<td>0.85</td>
</tr>
<tr>
<td>γ-HCH</td>
<td>1.0</td>
<td>1.0</td>
<td>1.0</td>
<td>1.0</td>
</tr>
<tr>
<td>HCB</td>
<td>1.0</td>
<td>1.0</td>
<td>1.0</td>
<td>1.0</td>
</tr>
<tr>
<td>TCDD-eqv (PCDD/PCDF)</td>
<td>-</td>
<td>0.98</td>
<td>0.99</td>
<td>1.0</td>
</tr>
<tr>
<td>PFOS</td>
<td>1.0</td>
<td>-</td>
<td>-</td>
<td>1.0</td>
</tr>
<tr>
<td>BDE-47</td>
<td>0.79</td>
<td>-</td>
<td>0.99</td>
<td>-</td>
</tr>
</tbody>
</table>
9 Pollutant regulation: conventions and legislation

9.1 The Stockholm Convention on Persistent Organic Pollutants
The Stockholm Convention on Persistent Organic Pollutants (POPs) is an international agreement requiring measures for reducing or preventing release of dangerous substances into the environment. The Stockholm Convention was adopted in 2001 and entered into force in 2004. The convention deals with organic compounds that are persistent and remain in the environment for a long time, have a potential for long-range transport, bioaccumulate in fatty tissues of organisms, and have adverse effects on human health or the environment. Initially, 12 chemicals were included in the treaty in 2001 (aldrin, chlordane, DDT, dieldrin, endrin, heptachlor, mirex, toxaphene, PCB, hexachlorobenzene, polychlorinated dibenzo-p-dioxins, polychlorinated dibenzofurans). In May 2009, an amendment was adopted into the convention, and nine additional chemicals were listed as POPs (hexabromodiphenylether, tetra-/pentabromodiphenylether, chlordecone, hexabromobiphenyl, lindane, α- and β-hexachlorocyclohexane, pentachlorobenzen and PFOS).

In May 2011 an amendment was adopted into the convention and technical endosulfan and its related isomers were added to the list with specific exemptions. Since November 2014, hexabromocyclododecane (HBCDD) is also included in the Stockholm Convention. Six more substances have been nominated to be included on the list, and are currently under review by the Persistent Organic Pollutants Review Committee; decabromodiphenyl ether (commercial mixture, “c-decaBDE”), dicofol, short-chained chlorinated paraffins (SCCPs), chlorinated naphthalenes, hexachlororbutadiene and pentachlorophenol (SC, 2008).

9.2 The Convention on Long-Range Trans boundary Air Pollution
The Convention on Long-Range Trans boundary Air Pollution (CLRTAP) was initiated in 1972 at a United Nations Conference on the Human Environment in Stockholm. After the scientific findings that acidification in Swedish lakes was caused by sulphur emission from continental Europe, the necessity for international measures to reduce emissions to air that had environmental effects far from its source, was addressed. In 1979, the convention was signed in Geneva, and entered into force in 1983. Initially, the convention focused on sulphuric compounds causing acidification, but later eight protocols were added for other groups of substances e.g., nitrogen oxides, volatile organic compounds (VOCs) and persistent organic pollutants (POPs) (http://www.unece.org/env/lrtap/lrtap_h1.htm).

9.3 EU chemical legislation

9.3.1 REACH
REACH is the EU chemicals policy that entered into force on the 1st of June 2007 (The European Parliament and The European Council, 2006). REACH stands for Registration, Evaluation, Authorization and Restriction of Chemical Substances. The policy places more responsibility on industry, and importers and users have to gather information about their chemicals, which they then report to the European Chemicals Agency (ECHA) based in Helsinki. ECHA manages REACH by gathering information and keeping databases of chemicals used in the EU (The European Commision, 2007).
9.3.2 RoHS directive

The Directive on the Restriction of Hazardous Substances (RoHS) was adopted in February 2003. The RoHS directive reduces the use of six chemical substances in electrical or electronic products that were released on the market after July 2006. These substances are mercury, cadmium, lead, chromium VI, polybrominated biphenyls and polybrominated diphenyl ethers. The maximum allowed amount of these substances (based on weight) is 0.01% for cadmium, and 0.1% for the other substances (KEMI, 2011b).

9.3.3 Water Framework Directive

The Water Framework Directive (WFD) aims to achieve good ecological and chemical status of all surface waters and ground water bodies in the EU by 2015. The WFD was adopted in October 2000, and deals with fresh water, coastal-zone and estuary waters. Within the WFD, a list of 33 prioritized substances has been established, and eight additional substances were later added. To evaluate if “good chemical status” has been achieved, threshold values, or Environmental Quality Standards (EQS), have been established for the listed substances (see chapter 10). It is the responsibility of each member state to assess and report if the goal has been fulfilled (The European Commision, 2000).

9.4 Swedish chemical legislation

One of the 16 Swedish environmental quality objectives is “A non-toxic environment”, which means that concentrations of non-naturally occurring substances should be close to zero, and naturally occurring substances should be close to background concentrations. Their impact on human health and ecosystems should be negligible (KEMI, 1999). The agency responsible for coordinating this work is the Swedish Chemicals Agency (KEMI). The Swedish chemical legislation follows EU legislation. Much of the national legislations that existed before June 2007 were replaced by REACH. (KEMI).
10 Target levels for chemical status assessment

Good Environmental Status (GES), in accordance with the Marine Strategy Framework Directive 2008/56/EC (MSFD), is defined as “concentrations of contaminants at levels not giving rise to pollution effects” (OJEC No. L164 25.6.2008, 2008). GES is determined from quality assessments based on target levels representing a threshold that should not be exceeded. Established to protect sensitive organisms from the harmful effects of hazardous substances, target levels have been developed within several groups or conventions; Environmental Quality Standards (EQS) developed within the EC to evaluate GES, and the Environmental Assessment Criteria (EAC), developed within OSPAR. In addition to EQSs and EACs, chemical status can also be assessed from the point of human consumption. Maximum levels for contaminants in food are set in Commission Regulation No 1881/2006 (OJEC No. L364 20.12.2006, 2006).

The Environmental Quality Standards Directive (2013/39/EU) lays Environmental Quality Standards (EQSs) and for priority substances and certain other pollutants, as provided for in Article 16 of the WFD, with the aim of achieving good surface water chemical status (OJEC No. L226 24.8.2013, 2013). The objective is to protect pelagic and benthic freshwater and marine ecosystems, as well as human beings from adverse impacts of chemical contaminants. The annual average concentration (AA-EQS) refers to the annual arithmetic mean concentration providing protection against chronic exposure and cover long-term chemical effects in biota. For some bioaccumulation substances EQS for biota have been set (EQSbiota). The methodological framework used in deriving these EQSs is described in Common Implementation Strategy for the Water Framework Directive (COM(2011) 876 31.1.2012, 2011, EC, 2011). Substance EQS Data Sheets (SDS) contains background information regarding the development of EQS, available at the CIRCABC webpage (CIRCABC). Here, data from ecotoxicological studies are compiled to Specific Quality Standards (QS), derived for water, sediment, biota (QSbiota), which is set to protect for secondary poisoning in predators, and human health (QShh). To date, EQSbiota is available for 11 out of 45 substances.

Within the OSPAR convention, Environmental Assessment Criteria (EAC) has been developed for interpretation of chemical monitoring data in sediments and biota (OSPAR Commission, 2014). Concentrations below the EACs are considered to present no significant risk to the environment and may be considered as related to the EQSs. In this report, primarily internationally agreed target levels such as EQS, EAC or EC recommendations for foodstuffs are used. If reliable target levels have been produced with specific regard to Swedish environmental conditions, these are considered (e.g. HCH and BDEs). Only one type of target level is applied within each substance group (e.g. we do not mix EQS and EACs depending on availability of different PCB congeners). Concentration of substances lacking internationally agreed target levels are presented without evaluation against target levels (e.g chromium, copper, zinc, arsenic, silver, aluminium, tin and bismuth).
Table 10.1. Target levels for various environmental pollutants.

<table>
<thead>
<tr>
<th>Group of substance</th>
<th>Target levels</th>
<th>Background reference</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Fish (µg/kg ww)</td>
<td>Mussels (µg/kg dw)</td>
</tr>
<tr>
<td>Metals</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Cadmium</td>
<td>160*</td>
<td></td>
</tr>
<tr>
<td>Lead</td>
<td>300</td>
<td>1500 ww</td>
</tr>
<tr>
<td>Mercury</td>
<td>20</td>
<td></td>
</tr>
<tr>
<td>Nickel</td>
<td>1230*</td>
<td></td>
</tr>
<tr>
<td>Pesticides</td>
<td></td>
<td></td>
</tr>
<tr>
<td>DDE (p,p')</td>
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<td></td>
</tr>
<tr>
<td>HCH (incl. lindane)</td>
<td>2.6/26</td>
<td></td>
</tr>
<tr>
<td>PCBs</td>
<td></td>
<td></td>
</tr>
<tr>
<td>CB-28</td>
<td>64 lw</td>
<td></td>
</tr>
<tr>
<td>CB-52</td>
<td>108 lw</td>
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<td>120 lw</td>
<td></td>
</tr>
<tr>
<td>CB-118</td>
<td>24 lw</td>
<td></td>
</tr>
<tr>
<td>CB-138</td>
<td>316 lw</td>
<td></td>
</tr>
<tr>
<td>CB-153</td>
<td>1600 lw</td>
<td></td>
</tr>
<tr>
<td>CB-180</td>
<td>480 lw</td>
<td></td>
</tr>
<tr>
<td>Other</td>
<td></td>
<td></td>
</tr>
<tr>
<td>ΣPBDEs (BDE-28, -47, -99, -100, -153, and -154)</td>
<td>0.0085</td>
<td>EQSbiota</td>
</tr>
<tr>
<td>HBCDD</td>
<td>167</td>
<td></td>
</tr>
<tr>
<td>ΣPCDDs+PCDFs</td>
<td>0.0035</td>
<td></td>
</tr>
<tr>
<td>ΣPCDDs+PCDFs+dl-PCBs</td>
<td>0.0065</td>
<td>0.0012</td>
</tr>
<tr>
<td>HCB</td>
<td>10</td>
<td></td>
</tr>
<tr>
<td>PFOS</td>
<td>9.1</td>
<td></td>
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</tbody>
</table>

*The target levels presented in the table for Cadmium and Nickel are based on QSbiota secondary poisoning, however secondary poisoning are not considered as the most sensitive protection objective for these substances and will thus not provide the highest protection level.

10.1 Metals

10.1.1 Cadmium

There is no EQSbiota or EAC developed for cadmium. The QSbiota secondary poisoning is set at 0.16 mg/kg prey tissue wet weight and evaluates whole fish concentrations in a freshwater system. The EC foodstuff regulation sets a maximum level for muscle meat at 0.05 mg/kg wet weight. The directive states that where fish are intended to be eaten whole, the maximum level shall apply to the whole fish.

Selected target level: QSbiota
10.1.2 Lead
There is no EQS_{biota} or EAC developed for lead. The QS_{biota} is set at 3600 µg/kg prey tissue wet weight and evaluates whole fish concentrations. The EC foodstuff regulation sets a maximum level for lead in muscle meat of fish at 300 µg/kg wet weight. The directive states that where fish are intended to be eaten whole, the maximum level shall apply to the whole fish.

*Selected target level: EC foodstuff regulation*

10.1.3 Mercury
The EQS_{biota} for mercury is set at 20 µg/kg (methyl mercury) prey tissue wet weight to protect against secondary poisoning. There is no EAC developed for mercury. The EC foodstuff regulation sets a maximum level for mercury at 0.5 mg/kg wet weight. The directive states that where fish are intended to be eaten whole, the maximum level shall apply to the whole fish.

*Selected target level: EQS_{biota}*

10.1.4 Nickel
There is no EQS_{biota} or EAC developed for nickel. The QS_{biota, secondary poisoning} is set at 12.3 mg/kg prey tissue wet weight. The QS_{hh} is set at 40.8 mg/kg fishery product wet weight. There is no EC foodstuff regulation developed for nickel.

*Selected target level: no target suitable*

10.2 Pesticides

10.2.1 DDTs, (DDT, DDE and DDD)
There are no EQS or EC foodstuff regulation developed for any of the DDTs. The EAC developed for DDE is set at 0.005 mg/kg wet weight.

*Selected target level: EAC*

10.2.2 HCH
There are no EQS or EC foodstuff regulation developed for HCHs. The EACs developed for γHCH in fish liver is set at 11 µg/kg lipid weight. With regard to Swedish levels of organic carbon in the sediments and factors for bioconcentration (BCF) and biomagnification (BMF), the Swedish Environmental Research Institute (IVL) have performed translations between EQS for surface water to biota (Lilja et al. 2010). The IVL target level is set for the sum of HCH (including lindane) at 26 µg/kg wet weight in a limnic environment and 2.6 µg/kg wet weight in a marine environment.

*Selected target level: IVL*

10.3 PCBs
Selected target level: EAC

10.4 Brominated flame retardants

10.4.1 BDEs
The EQS\textsubscript{biota} for sumBDEs, based on protection of human health, is set to a concentration at 0.0085 µg/kg wet weight. The QS\textsubscript{biota secondary poisoning} are set at 44 µg/kg wet weight. There are no EAC or EC foodstuff regulation developed for BDEs.

Selected target level: EQS\textsubscript{biota}

10.4.2 HBCDD
The EQS\textsubscript{biota} for concentrations of HBCDD is based on secondary poisoning of predators and set at 167 µg/kg fresh weight. There are no EAC or EC foodstuff regulation developed for HBCDD.

Selected target level: EQS\textsubscript{biota}

10.5 Other

10.5.1 Dioxins, furans and dioxin-like PCBs.
The EQS\textsubscript{biota} for concentrations of dioxins, furans and dioxinlike PCBs is based on the EC foodstuff regulation and set at 0.0065 µgWHO05-TEQ /kg ww. The QS set for human health was identified as the critical EQS because of the consensus regarding the value used in existing food legislation and because there is a greater uncertainty regarding the values calculated for QS secondary poisoning (0.0012 µgWHO05-TEQ /kg ww) The EC foodstuff regulation for only dioxins and furans is 0.0035 µgWHO05-TEQ /kg ww.

Selected target level: EQS\textsubscript{biota}

10.5.2 HCB
The EQS\textsubscript{biota} for HCB is based on human health and set at 10 µg/kg fishery product wet weight. There is no EC foodstuff regulation developed for HCB.

Selected target level: EQS\textsubscript{biota}

10.5.3 PFOS
The EQS\textsubscript{biota} for concentrations of PFOS is based on human health and set at 9.1 µg/kg wet weight while the QS\textsubscript{biota for protection via secondary poisoning} is set at 33 µg/kg ww. There are no EAC or EC foodstuff regulation developed for PFOS.

Selected target level: EQS\textsubscript{biota}
11 Biological variables

11.1 Results

11.1.1 Spatial Variation

Figure 11.1. Spatial variation in mean fat percentage in perch muscle.
Figure 11.2. Spatial variation in mean age (year) in perch.

Figure 11.3. Spatial variation in mean total length (cm) in perch.
Fat content in perch muscle is relatively consistent in all lakes sampled within the program, however, it seemed to be somewhat lower in the southern part of Sweden (around 0.5%). The highest fat percentage was found in perch muscle from Lake Spjutsjön (0.78%) in the County of Dalarna and the lowest (0.50% and 0.52% respectively) in Lake Sännen in the County of Blekinge and in Lake Krageholmsgssjön in the County of Skåne (Fig. 11.1).

The fat content in pike muscle from Lake Bolmen and Lake Storvindeln are of similar magnitude (with a mean value of approximately 0.6 % fat over the monitored time period). The fat content in Arctic char muscle is higher than for pike and perch with a mean fat content of approximately 1.5 % in char from both of the lakes (Fig. 11.5).

The age of the perch sampled within the programme is lower in the most southern and south eastern parts of Sweden (Fig. 11.2).

The total length of the perch sampled is very homogenous, which is a result of the perch sent for analyses are selected within a certain length span (Fig.11.3).

Some variation is seen for the total weight but no clear spatial pattern is observed (Fig. 11.4).

### 11.1.2 Temporal variation

Fat content in pike from Lake Bolmen and Lake Storvindeln shows significant downward trends during the time period 1967–2015 and 1970-2015 (Fig. 11.5), annual mean decrease of
1.0% and 0.59% respectively. The fat content in Arctic char from Lake Tjulträsk shows a significant upward trend (2.0% per year) during the whole monitoring period (1982–2015) but no trend could be detected in Arctic char from Lake Abiskojaure.

The fat content in perch from Lake Skärgölen (Fig. 11.6) shows a significant downward trend of 0.42% per year during the time period 1980–2015, but a significant upward trend of 1.3% per year during the last ten years. No trend is seen in perch from Lake Stensjön.

The number of years required to detect an annual change of 10% for fat content varies between 6–13 years for the pike, char and perch time series.

Figure 11.5. Fat content in Arctic char muscle (Lake Abiskojaure and Lake Tjulträsk) and in pike muscle (Lake Bolmen and Lake Storvindeln).
The fat content in pike from Lake Bolmen and Lake Storvindeln and in perch from Lake Skärgölen shows significant downward trends during the monitoring period. The fat content in Arctic char from Lake Tjulträsk is significantly increasing during the monitoring period (1982–2015) as well as in perch from Lake Skärgölen the last ten years.

The age of the perch sampled within the programme is somewhat lower in the most southern and south eastern parts of Sweden. Since the length of the fish is within the same length interval at all sampling sites, this indicates a faster growth rate in these areas with younger fish.

Figure 11.6. Fat content in perch muscle (Lake Skärgölen and Lake Stensjön).
12 Mercury – Hg

Updated 16.10.31

12.1 Introduction

12.1.1 Usage, Production and Sources
Mercury exists naturally in the environment in a number of chemical and physical forms. The main inorganic forms include Hg⁰ (metallic), Hg₂⁺ (mercurous), Hg³⁺ (mercuric). Organic forms include CH₃HgCH₃ (dimethylmercury) and CH₃Hg⁺ (monomethylmercury) (Suzuki T. et al., 1991).

Some of the more well-known uses of mercury include thermometers, barometers, sphygmomanometers (blood pressure cuffs), float valves (e.g., ball cock in flushing system of toilets), some electrical switches, amalgam for dental restoration, batteries, fluorescent lamps, anti-lock braking systems (ABS) in some 4WD vehicles and airbag sensors in some vehicle models. It can also be found in beauty products, such as mascara, as thiomersal. For a comprehensive list of mercury usage in everyday life, see Huber (Huber, 1997). Highly toxic and bioaccumulatory methylmercury compounds were previously used as fungicides and were produced as unwanted byproducts of the chemical industry (Clarkson, 1993).

Natural sources of mercury include volcanoes, forest fires, fossil fuels and cinnabar ore, which is mined primarily in Spain and Italy, although shortages of this rare metal have encouraged mining in other countries (Calvert, 2007). There are numerous atmospheric anthropogenic sources of mercury such as fossil fuel combustion, mining, smelting and solid waste combustion. Anthropogenic sources to soil and water include fertilisers, industrial wastewater disposal, landfills, the manufacture of cement and metals, and through other industrial processes. In Sweden, a south to north gradient exists in atmospheric mercury concentration, due to the south being closer to source points in Europe (Wångberg I and Munthe J., 2001). However, mercury use has almost ceased in Sweden (AMAP/UNEP, 2008).

In Swedish top layer soils (mor), the highest mercury concentrations are seen in the south, decreasing towards the north, with considerable local variation. Mercury concentrations vary regionally, with means from 0.2 mg/kg to 0.5 mg/kg seen. Natural background levels in mor/top layer soils are estimated to be 0.07 mg/kg, based on concentrations seen from the least affected northern areas. Natural background mercury concentrations in pike are estimated to be 0.2 mg/kg (The European Commision, 2001).

12.1.2 Environmental Fate
Mercury concentration in fish is highly correlated with water pH and acidic conditions favouring mercury methylation as well as increased water temperature, which also is known to increase methylation rates (Doetzel, 2007). Sulphate reducing bacteria has been shown to be a controlling factor of mercury methylation in estuarine sediments (Choi and Bartha, 1994). Fish biology also influences mercury levels, with age, size and diet affecting bioaccumulation rates (Doetzel, 2007).
12.1.3 Toxic Effects
Mercury is known to bioaccumulate (Clarkson, 1993). Methylmercury is the form of mercury of most concern to human health and ecosystem processes. Methylmercury combines with the amino acid cysteine to form a structure similar to another amino acid, methionine, which penetrates all mammalian cells and easily crosses the blood-brain barrier, from whence the central nervous system can be affected (Suzuki T. et al., 1991, Huber, 1997). High exposure can affect brain development, with young children and infants most at risk (Doetzel, 2007), as methylmercury disturbs cell division and therefore development (Huber, 1997).

The severity of symptoms after mercury exposure depends upon exposure level. Symptoms related to severe exposure are well documented after two major disasters of methylmercury contamination in Iraq in 1972, and Japan in 1957 (for a brief overview see (Amin-Zaki L. et al., 1974, Rustam and Hamdi, 1974, Clarkson, 1993, Huber, 1997). Symptoms are related to type of exposure, for example, inhalation of elemental mercury vapours results in respiratory problems, followed by neurological disturbance and general systemic effects. However, one of the most common routes of mercury exposure is via ingestion of methylmercury (Ratcliffe et al., 1996), often through consumption of contaminated fish (Huber, 1997), the risk of which can be greater for in utero children in pregnant women (Koren and Bend, 2010). Exposure becomes problematic if contaminated fish (or other contaminated foodstuffs) are eaten often, and neurological effects in both adults and children in utero can be seen (Ratcliffe et al., 1996).

Wildlife in all environments is affected by mercury accumulation; however animals in aquatic systems appear to show more intense bioaccumulation/biomagnification effects than terrestrial species (Huber, 1997). Bioaccumulation usually occurs through diet (Huber, 1997). A biomagnification effect is seen in fish at higher trophic levels compared to those at lower trophic levels (Da Silva et al., 2005). In the 1960s, the usage of methylmercury compounds as fungicides on seed grains led to the realisation that this compound was an ecological poison. The connection was made since birds of prey that predated on smaller birds that in turn had fed on methylmercury treated grains, suffered from severe population declines and a high mercury load (Clarkson, 1993). While methylmercury accumulates in fish muscle, the highest concentrations are generally seen in the blood, spleen, kidney and liver. In mammals and birds, the highest concentrations are typically seen in the feathers and fur. Embryos and very young animals tend to be the most affected by mercury damage due to its ability to interfere with cell division processes (Huber, 1997).

12.1.4 Conventions, aims and restrictions
The Minister Declaration from 1988, within HELCOM, calls for a reduction of the discharge of mercury to air and water by 50% by 1995, with 1987 as the base year (HELCOM, 1988).

The use of alkylmercury compounds for use in the agriculture as well as in the paper pulp industries has been banned in Sweden since 1966 (KEMI, 2010).

During the 1990s the production and selling of certain products containing mercury, e.g. thermometers and other measuring instruments, were banned. In 1997 it was forbidden to export mercury and mercury compounds to other countries. In 2009 the government adopted a general ban against all mercury in Sweden (KEMI, 2015). In 2011 within EU it became forbidden to export mercury.
12.1.5 Target Levels
The target level (TL) used for mercury in the time series for fish is 20 µg/kg wet weigh. The original target level is set for concentrations of mercury in whole-fish and the data presented in this report is measured in muscle. However, in a study on conversion factors between muscle and whole-fish (Faxneld S. et al., 2015) the existing target level was recalculated to the corresponding muscle value, and a value of 21 µg/kg wet weigh was derived. But, since this value is very close to the original target level, and probably within the margin of error, 20 µg/kg wet weigh is used. For further information on TL and selection of target level see chapter 10.

12.2 Results
12.2.1 Spatial Variation

![Hg, perch muscle](image)

**Figure 12.1.** Spatial variation in concentration (ng/g wet weight) of Hg in perch muscle.
The highest concentrations of mercury were found in perch from Tärnan (359 ng/g wet weight) and Stora Envättern (353 ng/g wet weight) in the County of Stockholm, and Lake Fräcksjön (335 ng/g wet weight) in the County of Västra Götaland in 2013–2015. The lowest concentrations of mercury (60 ng/g wet weight) in perch muscle were from Lake Krageholmssjön in the County of Skåne and Lake Fysingen (64 ng/g wet weight) in the County of Stockholm (Fig. 12.1).

The concentration of mercury in pike from Lake Bolmen (mean 360 ng/g wet weight) and Lake Storvindeln (mean 300 ng/g wet weight) is more than ten times higher than in Arctic char from Lake Abiskojaure (Fig. 12.2).

12.2.2 Temporal variation

The time series for Arctic char from Lake Abiskojaure show a significant downward trend for mercury of 0.62% per year for the whole time period (Fig. 12.2). No significant trend is detected for mercury in pike from Lake Bolmen and Lake Storvindeln or in the majority of the perch time series (Fig. 12.2–12.5). However, in Lake Övre Skärsjön a significant upward trend of 4.9% are seen during the most recent ten years.

In addition, there is a significant non-linear trend for the mercury concentration in both pike from Lake Storvindeln as well as in perch from Lake Skärgölen. In both cases the concentration is lower in the most recent years compared to fifteen years ago.

The number of years required to detect an annual change of 10% for mercury varies between 8–17 years for the pike, char and perch time series.

Figure 12.2. Mercury concentrations (ng/g fresh weight) in Arctic char muscle (Lake Abiskojaure) and in pike muscle (Lake Bolmen and Lake Storvindeln). The green area denotes the levels below the suggested target value for mercury in fish.
Figure 12.3. Mercury concentrations (ng/g fresh weight) in perch muscle from Lake Bysjön, Lake Stora Envättern and Lake Skärgölen. The green area denotes the levels below the suggested target value for mercury in fish.
Figure 12.4. Mercury concentrations (ng/g fresh weight) in perch muscle from Lake Fiolen, Lake Hjärtsjön and Lake Krageholmssjön. The green area denotes the levels below the suggested target value for mercury in fish.

Figure 12.5. Mercury concentrations (ng/g fresh weight) in perch muscle from Lake Remmarsjön, Lake Degervattnet, Lake Stensjön and Lake Övre Skärsjön. The green area denotes the levels below the suggested target value for mercury in fish.
12.2.3 Comparison to thresholds
The suggested target level for mercury based on ECs EQS\textsubscript{biota} (Environmental Quality Standard) of 20 ng/g wet weight was exceeded in all species and lakes sampled within the programme (Fig. 12.2–12.6).

Figure 12.6. Spatial variation in concentration (ng/wet weight) of mercury in perch muscle. The green sections of the bars are representing concentrations under the threshold level (20 ng/g wet weight) and the red sections concentrations above.

12.3 Summary
The highest concentration of mercury in perch (2013–2015) was found in Lake Tärnan (359 ng/g wet weight) and Lake Stora Envättern (353 ng/g wet weight) in the County of Stockholm, and Lake Fräcksjön (335 ng/g wet weight in the County of Västra Götaland.

Mercury concentrations vary between species and sites. Seen over time, the mercury concentration has decreased in Arctic char from Lake Abiskojaure since the end of the 1970s. On the other hand perch from Lake Övre Skärsjö shows increasing concentrations during the last ten years. For the other lakes and species, no clear trends are observed.

In all areas and species mercury concentration is above the suggested target level.
13 Lead – Pb

Updated 16.10.31

13.1 Introduction

13.1.1 Usage, Production and Sources
Lead occurs in many isotopes, but only three are stable. There are four natural isotopes, $^{204}\text{Pb}$, $^{206}\text{Pb}$, $^{207}\text{Pb}$ and $^{208}\text{Pb}$. $^{204}\text{Pb}$ is slightly radioactive, and has a half life of 22.2 years. In nature, lead is usually found in ore with zinc, silver or copper. Atmospheric sources of lead in Sweden show a south to north gradient, due to northward atmospheric transport from sources located elsewhere in Europe (Renberg et al., 2000). The main sources of lead pollution in Sweden come from ammunition, historic emissions from leaded petrol and associated contamination in road side soils (eventhough leaded gasoline was eliminated in 1994 in Sweden (Faiz et al., 1996), lead pigments, cables and batteries. There are also point sources (e.g. metal works) that have resulted in high local pollution (Bergbäck et al., 1992), e.g. a secondary lead smelter in Landskrona where lead from car batteries is recycled (Farago et al., 1999).

The primary use for lead is in car batteries. There are numerous other uses including, but not limited to, lead in the ballast keel of sailboats, scuba diving weight belts, fishing sinkers, firearms (bullets and shot), colouring elements in paints and ceramic glazes, PVC plastics, lead sheeting used for sound proofing, lining chemical treatment baths, storage vessels, roofing, cladding, organ pipes, soldering, electrodes, high voltage power cables, tennis racquets, statues, sculptures, anti-knocking additive in aviation fuel, leaded gasoline, solar energy cells and infrared detectors and coffins. Houses built prior to 1980 are at a higher risk of having been painted with lead-based paints. Many cities did (and some still do) use lead water and sewage pipes. Lead can leach out of the water pipes into drinking water. Lead arsenate was the most commonly used insecticide in deciduous fruit tree orchards prior to the introduction of DDT in 1947. High lead levels are still found in some top soils in the USA (Peryea and Creger, 1994, Peryea and Kammereck, 1997).

13.1.2 Environmental Fate
Increased acidity levels appear to contribute to increased lead bioavailability in soils (Jin et al., 2005). In lakes, the level of lead in fish body tissues is often greater in low-alkalinity water compared to lakes with a higher pH (Spry and Wiener, 1991). These results indicate that pH may influence lead bioavailability in water systems and sediments.

13.1.3 Toxic Effects
Lead is a non-essential element (Tewari et al., 1987) and a known neurotoxin, damaging the nervous system as well as causing brain and blood disorders. The toxic effects of lead involve several organ systems and biochemical activities. The risk is highest for children and the unborn, partly because of high permeability of lead across both the blood-brain barrier and the placenta (Klaassen C.D. and Rozman K., 1991). Some neurophysiological development affects can be seen in children even at low levels of lead exposure (Gidlow, 2004).

Lead is known to bioaccumulate in soft tissue, but to a much greater extent in the bone matrix. Approximately 90% of the total amount of lead in humans is found in the skeleton (Klaassen...
C.D. and Rozman K., 1991). Between 90% and 95% of the lead that is found in blood is distributed to the red blood cells, where haemoglobin synthesis can be inhibited, and subsequently symptoms such as anaemia can occur (Gidlöw, 2004). In females, lead is a known abortifacient but problems in male reproduction are equivocal (Gidlöw, 2004).

In animals, absorbed lead enters the blood and soft tissues but is eventually redistributed to the bones. In birds, lead shot is a common cause for lead poisoning (Cook and Trainer, 1966, Pattee et al., 1981), and there have been reports of fishing sinkers causing bird deaths (Locke et al., 1982). In Sweden, bird death from lead poisoning is more common in swans, geese and ducks, but has also been reported in woodpeckers. Lead levels were found to be highest in woodpecker liver and kidney (Mörner and Petersson, 1999).

13.1.4 Conventions, Aims and Restrictions
The Minister Declaration from 1988, within HELCOM, calls for a reduction of the discharges of lead to air and water by 50% by 1995, with 1987 as the base year (HELCOM, 1988).

The recommended limit for children’s food is set by the Swedish National Food Administration (SNFA) at 50 ng/g fresh weight (Livsmedelsverket, 1993). Within the European Community, the limit in fish muscle is set at 0.2 µg/g, and in mussels at 1.5 µg/g. (The European Commision, 2002).

13.1.5 Target Levels
The target level (TL) used for lead in the time series for perch is 1000 µg/kg wet weight in whole-fish. For further information on TL and selection of target level see chapter 10. The original TL has been recalculated to dry weight in liver for each time series to fit the presented data. The recalculation of the TL for whole-fish is based on a study in which concentrations of lead in the liver and whole-fish of perch from the limnic environment were compared (Faxneld S. et al., 20xx). After recalculating the existing target level of 1000 µg/kg wet weight in whole-fish the new derived target level in liver is 0.30 µg/kg wet weight. The recalculation to dry weight is based on the dry weight in each time series. The recalculated target level (Tv) together with the dry percentage (dp) is shown above the statistical information in each time series.
13.1 Results

13.1.1 Spatial Variation

![Pb, perch liver](image)

**Figure 13.1.** Spatial variation in concentration (µg/g dry weight) of lead in perch liver.

The highest concentrations of lead (0.17 µg/g dry weight) were found in perch liver from Lake Fiolen in the County of Kronoberg and Lake Sännen in the County of Blekinge in (2013–2015). The lowest concentration of lead (0.009 µg/g dry weight) was found in perch liver from Lake Krankesjön in the County of Skåne (Fig. 13.1).

The concentration of lead is of a similar magnitude in pike from Lake Storvindeln and Lake Bolmen and in Arctic char from Lake Abiskojaure (with concentrations around 0.007 µg/g dry weight in 2015) (Fig. 13.2).

13.1.2 Temporal variation

The pike and char time series show significant decreasing trends for lead of about 3.7–12% per year in all monitored lakes (Fig. 13.2) from 1969/1980/1998–2015. A significant decreasing trend is also observed in perch from Lakes Skärgölen, Fiolen, Hjärtsjön and Stensjön, of about 4.4-7.9%, 1997/2000–2015 (Fig. 13.3-13.5).

The decreasing trends for lead are consistent with other time series reported for lead in marine biota (Bignert A. et al., 2010), and are most probably a result of the ban of lead in gasoline in 1994.

The number of years required to detect an annual change of 10% for lead varies between 11–18 years for the pike, char and perch time series.
Figure 13.2. Lead concentrations (µg/g dry weight) in Arctic char liver (Lake Abiskojaure) and in pike liver (Lake Bolmen and Lake Storvindeln).

Figure 13.3. Lead concentrations (µg/g dry weight) in perch liver from Lake Bysjön, Lake Stora Envättern and Lake Skärgölen. The green area denotes the levels below the suggested target value for lead in fish.
Figure 13.4. Lead concentrations (µg/g dry weight) in perch liver from Lake Fiolen, Lake Hjärtsjön and Lake Krageholmsjön. The green area denotes the levels below the suggested target value for lead in fish.

Figure 13.5. Lead concentrations (µg/g dry weight) in perch liver from Lake Remmarsjön, Lake Degervatnet, Lake Stensjön and Lake Övre Skärsjön. The green area denotes the levels below the suggested target value for lead in fish.
13.1.3 Comparison to thresholds
In all areas, lead concentration in perch is below the derived target level of 0.30 µg/kg wet weight for liver, based on the QSbiota of 1 µg/g wet weight for whole-fish.

13.2 Summary

The highest concentration of lead in perch liver was found in Lake Fiolen in the County of Kronoberg and Lake Sännen in the County of Bleking in 2013–2015.

Lead concentrations varied between species and sites; however temporally, lead concentration has decreased by approximately 4–12% per year in the freshwater environment since the end of the 1960s.

In all areas, lead concentration in perch is below the suggested target level.
14 Cadmium – Cd

Updated 16.10.31

14.1 Introduction

14.1.1 Usage, Production and Sources
Cadmium is an important metal in many industrial applications. It was used extensively until the end of the 1970s in electroplating or galvanising because of its noncorrosive properties. Cadmium has also been used (and to some extent is still used) as a cathode material for nickel-cadmium batteries, and as a colour pigment for paints and plastics. However, its industrial use has decreased considerably during recent years. Sweden was the first country in the world to introduce a principal ban for the use of cadmium in certain industrial applications in 1982. Cadmium can also enter the environment as a by-product of zinc and lead mining and smelting, and as a byproduct in fertilisers.

Cadmium occurs in ores with zinc, copper and lead, and volcanic activity can increase atmospheric cadmium concentrations (Godt et al., 2006). In Sweden, cadmium discharge into the environment has been estimated to have decreased by approximately 45% in the years 1985–1990, while the airborne cadmium load during the same period is estimated to have decreased by approximately 15% (Notter, 1993).

14.1.2 Environmental Fate
A decreased cadmium load to the environment may cause a corresponding decrease in the amount of bioavailable cadmium. It has been shown that the mobility of Cd-ions has increased due to acidification, and that this may cause increased cadmium concentration in run off waters (Borg et al., 1989). Once absorbed, cadmium can be bound to metallothionein (MT) (Da Silva and Williams, 1994). It is known that various compounds can induce (and possibly also inhibit) the formation of MT in fish liver (Bouquegneau et al., 1975). A change in the amount of MT, due to induction, inhibition or ceased induction or inhibition, might thus change the metal concentration in the analysed liver tissue.

14.1.3 Toxic Effects
The most common source of cadmium for humans is via cigarette smoke. There is also a low risk of being exposed to cadmium via oral and dermal pathways. Cadmium is generally found in the liver or kidneys, 30% of the cadmium body burden is found in kidneys, with kidneys being the main organ for long term cadmium accumulation in humans. This can lead to renal tube dysfunction. Bone tissues are affected secondarily (after kidneys). At very high exposure rates, effects on the respiratory system (e.g. emphysema) are known, while the nervous system in developing animals appears to be sensitive. There have been some effects on reproduction, and some proof of carcinogenic effects. Cadmium transported in blood plasma becomes bound to albumin and is then preferentially taken up by the liver, where metallothionein is synthesised. The placenta is only a partial barrier to foetal exposure. Cadmium is excreted in faeces and urine (Godt et al., 2006).
14.1.4 Conventions, Aims and Restrictions

The Minister Declaration from 1988, within HELCOM, calls for a reduction in discharges of cadmium to air and water by 50% by 1995, with 1987 as the base year (HELCOM, 1988).

The Swedish Parliament has agreed on a general reduction of cadmium discharge, aiming at a reduction of 70% between 1985 and 1995, and further, that all use of cadmium associated with a risk of discharge to the environment, in a long term perspective, will cease (Sveriges Regering, 1991). Further legislative restrictions concerning cadmium in Sweden are; in 1982, the use of cadmium in electroplating and as a thermal stabiliser was banned in Sweden, in 1987, a fee on batteries containing cadmium was introduced in Sweden. This fee was raised considerably in 1991 and in 1993, the content of cadmium in fertilisers was restricted to 100g/ton of phosphorus. Since 2009, Sweden has followed the cadmium restrictions in fertilizers within REACH.

14.1.5 Target Levels

The target level (TL) used for cadmium in the time series for perch is 160 μg/kg wet weight in whole-fish. For further information on TL and selection of target level see chapter 10. Since cadmium data presented here are expressed as dry weight in liver, the original TL has been recalculated to represent the corresponding concentration in liver for comparison. The recalculation of the TL for whole-fish is based on a study that compared concentrations of cadmium in the liver and whole-fish of perch from the limnic environment (Faxneld S. et al., 20xx). The new derived target level for liver was 6.65 μg/g wet weight. The recalculation to dry weight is based on the dry weight in each time series. The recalculated target level (Tv) together with the dry weight percentage (dp) is shown above the statistical information in each time series.
14.2 Results

14.2.1 Spatial Variation

The highest concentration of cadmium (21 μg/g dry weight) was found in perch liver from Lake Sännä in the County of Blekinge (2013–2015). The lowest concentrations of cadmium in perch liver were seen in Lake Krageholmssjön (0.13 μg/g dry weight) and Lake Krankesjön (0.22 μg/g dry weight), both located in the County of Skåne (Fig. 14.1). The levels of cadmium in perch liver are generally somewhat higher in the south of Sweden than in the north.

The concentration of cadmium varies between species and sites (Fig. 14.1–14.5). In Arctic char the concentration of cadmium is more than four times higher (1.1 μg/g dry weight, mean over the whole time period) compared to pike from Lake Storvindeln and Lake Bolmen (0.05 and 0.28 μg/g dry weight respectively, mean over the whole time period) (Fig. 14.2).

14.2.2 Temporal variation

Significant downward trends were observed in perch liver from Lake Fiolen and Lake Stensjön, annual mean decrease of 4.5 and 3.6%, respectively, during 2000–2015 (Fig. 14.3 and 14.5). A downward trend was also indicated in perch from Lake Skärgölen (Fig. 14.3). The number of years required to detect an annual change of 10% for cadmium varies between 10–17 years for the pike, char and perch time series.
Figure 14.2. Cadmium concentrations (µg/g dry weight) in Arctic char liver (Lake Abiskojaure) and in pike liver (Lake Bolmen and Lake Storvindeln).

Figure 14.3. Cadmium concentrations (µg/g dry weight) in perch liver from Lake Bysjön, Lake Stora Envättern and Lake Skärgölen. The green area denotes the levels below the suggested target value for cadmium in fish.
Figure 14.4. Cadmium concentrations (µg/g dry weight) in perch liver from Lake Fiolen, Lake Hjärtsjön and Lake Krageholmssjön. The green area denotes the levels below the suggested target value for cadmium in fish.

Figure 14.5. Cadmium concentrations (µg/g dry weight) in perch liver from Lake Remmarsjön, Lake Degervattnet, Lake Stensjön and Lake Övre SKärsjön. The green area denotes the levels below the suggested target value for cadmium in fish.
14.2.3 Comparison to thresholds

In a few of the lakes (Fig 14.6) (Lakes Sännen, Stora Envättern, Limmingssjön, Övre Skärsviken, Fiolen, Hjärtesjön, Fräcksjön, Lilla Öresjön, Spjutsjön, Remmarsjön), cadmium concentration in perch is above the derived target level of 6.65 µg/g wet weight for liver based on the QSBiota of 160 µg/kg wet weight in whole-fish.

Figure 14.6. Spatial variation in concentration (µg/g dry weight) of cadmium in perch liver. The green sections of the bars are representing concentrations under the threshold level (6.65 µg/g wet weight in liver) and the red sections concentrations above.

14.3 Summary

The highest concentration of cadmium in perch liver was found in Lake Sännen in the County of Blekinge 2013–2015.

Cadmium concentrations varied between species and sites – downward trends were observed in two of the perch lakes while no trends were seen in all of the other lakes.

In 10 out of 27 lakes, cadmium concentrations in perch were above the suggested target level.
15 Nickel – Ni

15.1 Introduction

15.1.1 Usage, Production and Sources
The most common ores of nickel include pentlandite, pyrrhotite, and garnierite. In addition, nickel also occurs as an impurity in ores of iron, copper, cobalt, and other metals. Nickel occurs as a mixture of five isotopes, i.e. $^{58}\text{Ni}$, $^{60}\text{Ni}$, $^{61}\text{Ni}$, $^{62}\text{Ni}$, and $^{64}\text{Ni}$. Seven radioactive isotopes of nickel are also known; however, only $^{63}\text{Ni}$ of the radioactive isotopes is used in industrially.

The most important use of nickel is in the manufacture of a variety of alloys, including stainless steel (Cempel and Nikel, 2006). Moreover, nickel is also very popular in the battery industry. Nickel-cadmium and nickel-metal hydride batteries are the main line products. Nickel are used in a great variety of appliances, including hand-held power tools, compact disc players, pocket recorders, camcorders, cordless and cellular telephones, scanner radios, surge protectors electronic devices used for the detection of explosives and laptop computers. Nickel is also used in electroplating.

15.1.2 Environmental Fate
Nickel can be released to the environment both by natural sources and anthropogenic activities. Weathering of rocks and soils, volcanic emissions, forest fires are the main natural sources of atmospheric nickel. Anthropogenic activities producing nickel include combustion of fossil fuel, incineration of waste, using stainless steel utensils, smoking tobacco. Domestic wastewater effluents and non-ferrous metal smelter are responsible for the nickel contamination in aquatic ecosystems (Cempel and Nikel, 2006). In water, nickel can deposit to sediments or taken up by biota. Nickel typically accumulates in the surface soils once deposited, once released by industrial and agricultural activities (Scott-Fordsmand, 1997).

15.1.3 Toxic Effects
Nickel is one of the essential metals for the function of several animals, organisms, and plants. Nickel has not been recognized as a nutritional element for human. Therefore, exposure to nickel compounds can have adverse effects on human health. Human exposure to nickel is primarily through ingestion of contaminated drinking water or food and inhalation (Cempel and Nikel, 2006).

Allergic skin reactions caused by nickel have been reported as one of the most common causes of allergic contact dermatitis (Andrea, 2005). Erythema, eczema and lichenification can occur once skin is in contact with nickel. Nickel compounds have been shown as carcinogenic in some animals and modes of human exposure (World Health Organization, 1991, Kasprzak et al., 2003, Cempel and Nikel, 2006). Recent studies have reported the ability of nickel to enhance lipid peroxidation in the liver, kidney, lung, bone marrow and serum (Denkhaus and Salnikow, 2002, Cempel and Nikel, 2006).
15.1.4 Target Levels
The target level (TL) for nickel is 730 µg/kg wet weight for whole-fish. For further information on TL and selection of target level see chapter 10. The concentrations of nickel presented in this report are analysed in liver and according to a study (Faxneld S. et al., 20xx) where concentrations were compared between liver and whole-fish, no significant correlations were found. Thereby the available TL is not applicable on the results and thus not presented in the time-series for nickel.

15.2 Results
15.2.1 Spatial Variation

![Spatial variation in concentration (µg/g dry weight) of nickel in perch liver.](image)

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No general spatial pattern was observed for nickel in perch liver. The highest concentration of nickel (0.20 µg/g dry weight) was found in perch liver from Lake Fysingen close to Stockholm in 2013–2015. The lowest concentration of nickel (0.040 µg/g dry weight) in perch liver came from Lake Krageholmsjön in the County of Skåne (Fig. 15.1).

The concentration of nickel in Arctic char from Lake Abiskojaure with a mean concentration of 0.17 µg/g dry weight is three to four times as high as in pike from Lake Bolmen and Lake Storvindeln (Fig. 15.2). The nickel concentration in Arctic char is also higher compared to perch.
15.2.2 Temporal variation

The concentration of nickel in perch liver increased significantly, by 4.1–8.2% per year, in Lake Bysjön, Lake Stora Envättern, Lake Skärgölen and Lake Hjärtsjön (Fig. 15.3 and 15.4) from 2000–2015. In Lake Remmarsjön a significant increase (5.4% per year) was seen during the most recent ten years (Fig. 15.5). Moreover, a downward trend, of 6.9% annually, for the last ten years, was observed in perch from Lake Fiolen (Fig. 15.4). No significant trends were observed for the other matrices and sites within the programme (Fig. 15.2–15.5) for nickel. The number of years required to detect an annual change of 10% for nickel varies between 12–19 years for the pike, char and perch time series.

**Figure 15.2.** Nickel concentrations (µg/g dry weight) in Arctic char liver (Lake Abiskojaure) and in pike liver (Lake Bolmen and Lake Storvindeln).
Figure 15.3. Nickel concentrations (µg/g dry weight) in perch liver from Lake Bysjön, Lake Stora Envättern and Lake Skärgölen.

Figure 15.4. Nickel concentrations (µg/g dry weight) in perch liver from Lake Fiolen, Lake Hjärtsjön, and Lake Krageholmssjön.
Figure 15.5. Nickel concentrations (µg/g dry weight) in perch liver from Lake Remmarsjön, Lake Degervatnet, Lake Stensjön, and Lake Övre Skärsjön.

15.3 Summary

The highest concentration of nickel in perch liver was found in Lake Fysingen close to Stockholm in 2013–2015.

Nickel concentrations varied between species and sites and showed an upward trend in 50% of the perch time series during the monitored period.
16 Chromium – Cr

Updated 16.10.31

16.1 Introduction

The method for chromium analysis was changed in 2004. The new method gives lower concentrations in general. The level of quantification is also lower with the new method.

16.1.1 Usage, Production and Sources

The abundance of chromium in the Earth's crust is about 100 to 300 ppm in rock (Domy, 2001). Chromium does not occur as a free element in the environment. Rocks or sediments present a wide range of chromium concentration whereas natural water contain quite small amounts (Richard and Bourg, 1991). Most chromium is produced from chromite (FeCr$_2$O$_4$). Chromium is used in the manufacturing of stainless steels, electroplating, leather tanning, pigments for inks and paints.

There are four naturally occurring isotopes of chromium: $^{50}\text{Cr}$, $^{52}\text{Cr}$, $^{53}\text{Cr}$, $^{54}\text{Cr}$ and seven known radioactive isotopes (Eisler, 1986). Oxidation states of chromium can vary from chromium(0) to chromium(VI), in which chromium(III) and chromium(VI) are the most stable and important species for natural aquatic systems (Richard and Bourg, 1991). Aqueous chromium is present as the Cr$^{3+}$ cation under reducing environments and pH lower than 3.6. In waters with a pH of 4-10, chromium exists as CrOH$^{2+}$, Cr(OH)$_2^+$ and Cr(OH)$_3$ with an increasing number of hydroxides counter ions with increasing pH. chromium(VI) is only present in oxidizing conditions and at pH over 10, as Cr(OH)$_4^-$ (Richard and Bourg, 1991).

16.1.2 Environmental Fate

Chromium can be transported between various environmental media and once present in the environment, it can be taken up by humans and other biota. Chromium is introduced to the environment mainly through anthropogenic activities rather than through weathering processes (Eisler, 1986). Chromium alloy, metal production, coal combustion, municipal incinerators, cement production, and cooling towers are responsible for major atmospheric emissions of chromium (Towill et al., 1978). The transformation and transport of chromium in the atmosphere are in association with aerosols. Chromium is removed from the atmosphere by both wet and dry deposition and reintroduced via resuspension of chromium-containing soil particles. In the aquatic environment, the major sources of chromium are atmospheric deposition, industrial activities (i.e. electroplating, metal finishing industries and waste water treatment plants) and, of subordinate importance, by weathering of natural rocks (Kimbrough et al., 1999). Chromium leaks to soil and sediment mainly from human activities, i.e. using chromium in phosphate fertilizers, chromium plating bath, ferrochromium slag (de Lopez Camelo et al., 1997).

16.1.3 Toxic Effects

The toxicity of chromium is regulated by its oxidation state, irrespective of its total concentration. Chromium(III) appears to be a nutrient for some plants and animals, including
humans; however, chromium(VI) has been reported to be toxic to bacteria, plants, and animals (Richard and Bourg, 1991). Environmental properties, i.e. hardness, temperature, pH, and salinity of water, in combination with biological factors, i.e. species, life stage, sensitivities of local population, determines the toxicity of chromium to aquatic biota (de Lopez Camelo et al., 1997). In addition, interaction effects of chromium with other contaminants, duration of exposure, and chemical form of chromium are also important factors. In general, chromium(III) is less toxic than chromium(VI) due to the latter’s high oxidizing potential and its easier membrane permeability (Eisler, 1986, Richard and Bourg, 1991). For sensitive aquatic biota, LC₅₀ of chromium (III) were from 2000 to 3200 ppb while chromium (VI) ranged from 445 to 2000 ppb (Eisler, 1986). Chromium is a trace element that has significant biological effects to the human body. Small amounts of chromium are necessary for plants and animals to metabolize glucose and synthesize amino acids and nucleic acids (Richard and Bourg, 1991). Chromium deficiency leads to diabetes-like symptoms in humans (Towill et al., 1978). At high level, chromium can cause nausea, skin ulcerations or lung cancer depending on exposure pathway and amounts of uptake.

16.1.4 Conventions, Aim, and restriction
The maximum chromium concentration in drinking water, recommended by the Commission of European Communities, the World Health Organization or the U.S. Environmental Protection Agency, is 50 µg/L (Richard and Bourg, 1991).

16.1.5 Target levels
No national target level for biota is agreed upon for chromium.
16.2 Results

16.2.1 Spatial Variation

![Spatial Variation Diagram]

**Figure 16.1.** Spatial variation in concentration (µg/g dry weight) of chromium in perch liver.

No general spatial pattern was observed for chromium in perch liver. The highest concentration of chromium was found in perch liver from Lake Allgjuttern (0.22 µg/g dry weight) in the County of Kalmar in 2013–2015. The lowest concentration of chromium found in perch liver came from Lake Krageholmsjön in the County of Skåne (0.042 µg/g dry weight) (Fig. 16.1).

The mean concentration of chromium in pike from Lake Storvindeln is almost twice as high as in pike from Lake Bolmen (Fig. 16.2).

16.2.2 Temporal variation

The time series for chromium show significantly decreasing concentrations in most species and sites monitored within the programme (Fig. 16.2–16.5), but this significant decrease is most probably caused by the change in method for chromium analysis in 2004. Due to this fact, any trend for the last ten years must be considered a more accurate description of the trend of chromium concentrations within the monitoring programme, compared to the whole time series. In 11 of the thirteen lakes there is a significant increase of the chromium concentration in the last ten years. This is seen in pike from the Lakes Storvindel and Bolmen, annual mean increase of 17 and 14 %, respectively, as well as in perch from the Lakes Bysjön, Stora Envättern, Skärgölen, Fiolen, Hjärtsjön, Krageholmssjön, Remmarsjön, Stensjön, and Övre Skärsjön with an annual mean increase of 9.2–24%.

Furthermore, an increase of the chromium concentration is indicated in arctic char from Lake Abiskojaure.
The number of years required to detect an annual change of 10% for chromium varies between 16–24 years for the pike, char and perch time series.

Figure 16.2. Chromium concentrations (µg/g dry weight) in Arctic char liver (Lake Abiskojaure) and in pike liver (Lake Bolmen and Lake Storvindeln).

Figure 16.3. Chromium concentrations (µg/g dry weight) in perch liver from Lake Bysjön, Lake Stora Envättern, and Lake Skärgölen.
Figure 16.4. Chromium concentrations (µg/g dry weight) in perch liver from Lake Fiolen, Lake Hjärtsjön, and Lake Krageholmssjön.

Figure 16.5. Chromium concentrations (µg/g dry weight) in perch liver from Lake Remmarsjön, Lake Degervattnet, Lake Stensjön, and Lake Övre Skärsjön.
16.3 Summary

The highest concentration of chromium in perch liver was found in Lake Allgjuttern in the County of Kalmar in 2013–2015.

Chromium concentrations showed a general decreasing trend in all matrices during the monitoring period, but this decrease is most probably caused by the change of method for chromium analysis in 2004. However, in the majority of the time series there were a significant increase of the chromium concentrations during the last ten years.
17 Copper – Cu

17.1 Introduction

17.1.1 Usage, Production and Sources
Copper is a nutritionally essential metal, and its concentration is regulated by homeostatic mechanisms. Free copper is effectively controlled by metallothionein synthesis (Da Silva and Williams, 1994) induced by copper itself or by other substances. Although copper is not believed to accumulate with continued exposure, changes found in biological tissues may still reflect changes in concentration of the ambient water.

Copper occurs naturally in rocks, soil, water, sediment and at low levels in air. In its metallic form, copper is used for example, plumbing, building wire, telecommunications, automotive electrical wiring and air conditioning systems (Dorsey et al., 2004). Copper compounds are used to create alloys such as brass and bronze. Other anthropogenic sources include road run off (Rice et al., 2002) and mining of copper ore. Copper compounds are commonly used in agriculture as fungicides, as well as in wood, leather and fabric preservatives, or for water treatment (Dorsey et al., 2004).

17.1.2 Conventions, Aims and Restrictions
The Minister Declaration from 1988, within HELCOM, calls for a reduction in the discharge of copper to air and water by 50% by 1995, with 1987 as the base year (HELCOM, 1988).

17.1.3 Target Levels
No national target level for biota concerning copper is agreed upon.

Average copper concentration in the earth’s crust is 50 ppm (Dorsey et al., 2004). The ‘background concentration at diffuse loading’ in blue mussels for copper is <10 µg/g dry weight, proposed by Knutzen and Skie (Knutzen J. and Skei J., 1992).

17.2 Results

17.2.1 Spatial Variation
No general spatial pattern was observed for copper in perch liver. The highest concentration of copper (72 µg/g dry weight) was found in perch liver from Lake Övre Skärsjön in the County of Västmanland in 2013-2015. The lowest concentration of copper (8.1 µg/g dry weight) was found in perch liver from Lake Fysingen in the County of Stockholm (Fig. 17.1).

The concentration of copper in pike from Lake Bolmen is almost twice as high as in pike from Lake Storvindeln (Fig. 17.2).

17.2.2 Temporal variation
A downward trend of 4.8% is observed for copper in perch liver from Lake Bysjön during 2000–2015 (Fig. 17.3). In addition, a significant downward trend for the last ten years is seen for perch at Lake Stensjön, with an annual mean decrease of 3.5% (Fig. 17.5). In contrast, an upward trend for copper is seen in Krageholmssjön with an annual mean of 2.0% (Fig. 17.4). No trends are detected for the other matrices and sites monitored within the programme (Fig. 17.3–17.5) for copper.

The number of years required to detect an annual change of 10% for copper varies between 8–15 years for the pike, char and perch time series.
Figure 17.2. Copper concentrations (µg/g dry weight) in Arctic char liver (Lake Abiskojaure) and in pike liver (Lake Bolmen and Lake Storvindeln).

Figure 17.3. Copper concentrations (µg/g dry weight) in perch liver from Lake Bysjön, Lake Stora Envättern, and Lake Skärgölen.
Figure 17.4. Copper concentrations (µg/g dry weight) in perch liver from Lake Fiolen, Lake Hjärtsjön, and Lake Krageholmssjön.

Figure 17.5. Copper concentrations (µg/g dry weight) in perch liver from Lake Remmarsjön, Lake Degervatnet, Lake Stensjön, and Lake Övre Skärsjön.
17.3 Summary

The highest concentration of copper in perch liver was found in Lake Övre Skärsjön in the County of Västmanland in 2013–2015.

No general trend is observed for copper during the monitored period.
18 Zinc – Zn

18.1 Introduction

18.1.1 Usage, Production and Sources
Zinc is a nutritionally essential metal naturally present in some foods. It is a biological requirement for many animals and plants (National Institutes of Health, 2011). Zinc concentration is regulated by homeostatic mechanisms. Hence, zinc is not believed to accumulate with continued exposure, but changes found in biological tissues may still reflect changes in concentration of the ambient water. Zinc occurs naturally in the environment, but most zinc originates from human activities such as mining, steel production and coal burning. In its pure form, anthropogenic sources of zinc can include use in steel and iron galvanization to prevent rust and corrosion; production of brass and bronze and zinc is also used to make dry cell batteries. Zinc compounds are used in industry for things such as making white paints and ceramics, producing rubber, preserving wood and dyeing fabrics (ATSDR, 2005). Tyre tread material contains approximately 1% weight of zinc. Wear of tyres on road surfaces can contribute a small amount of zinc to the environment (Councell T.B. et al., 2004). Some sunscreens use zinc oxide nanoparticles (Osmond and Mccall, 2010), other zinc compounds can be found in, for example, deodorants, nappy rash creams and anti-dandruff shampoos (ATSDR, 2005).

18.1.2 Environmental Fate
Zinc is present in water, air and soil. In air, zinc is present mostly as small particles that fall to the earth and drain into waterways with precipitation. Most of this zinc ends up settling in sediment at the bottom of water bodies. However, some zinc can remain bound to the soil. Zinc dissolved in water can increase the acidity (ATSDR, 2005).

18.1.3 Conventions, Aims and Restrictions
The Minister Declaration from 1988, within HELCOM, calls for a reduction in the discharge of zinc to air and water by 50% by 1995, with 1987 as the base year (HELCOM, 1988).

18.1.4 Target levels
No national target level for biota is agreed upon for zinc.

18.2 Results

18.2.1 Spatial Variation
Figure 18.1. Spatial variation in concentration (µg/g dry weight) of zinc in perch liver.

No general spatial pattern was observed for zinc in perch liver. The highest concentration of zinc (135 µg/g dry weight) was found in perch liver from Lake Sännen in the County of Blekinge in 2013–2015. The lowest concentration of zinc (92 µg/g dry weight) came from perch liver from Lake Svartsjön in the County of Södermanland (Fig. 18.1).

Pike in Lake Bolmen had almost twice as high concentration (197 µg/g dry weight) in 2012 compared to pike in Lake Storvindeln, Arctic char in Lake Abiskojaure and perch from the lakes in the monitoring programme (Fig. 18.2–18.5).

18.2.2 Temporal variation

A significant downward trend was seen for zinc in pike liver from Lake Storvindeln with an annual change of 1.2%. In perch liver a decrease was seen in a majority of the sampling sites with annual mean changes of about 0.93–1.9% (Fig. 18.2–18.5). In perch liver from Lake Krageholmsgssjön an upward trend was seen for the last ten years with an annual increase of 1.5%.

The number of years required to detect an annual change of 10% for zinc varies between 5–14 years for the pike, char and perch time series.
Figure 18.2. Zinc concentrations (µg/g dry weight) in Arctic char liver (Lake Abiskojaure) and in pike liver (Lake Bolmen and Lake Storvindeln).

Zn, µg/g dry w., liver

Abiskojaure, arctic char
n(tot)= 330, n(year)= 34
mean=14 (107, 122)
slope= -39%(-0.28)
CV(%)=1%, 1.1%, 6 yr
y(15)=107 (64, 0.122)
r=0.4, p=0.21

Storvindeln, pike
n(tot)= 430, n(year)= 44
mean=124 (115, 134)
slope= -12%(-1.2, -0.8)
CV(%)=1%, 1.3%, 6 yr
y(15)=107 (63.2, 1105)
r=0.45, p=0.001

Bolmen, pike
n(tot)= 180, n(year)= 18
mean=120 (120, 128)
slope= -7%(-0.02, 0.8)
CV(%)=1%, 1.3%, 4 yr
y(15)=107 (63.2, 1105)
r=0.45, p=0.02

Figure 18.3. Zinc concentrations (µg/g dry weight) in perch liver from Lake Bysjön, Lake Stora Envättern, and Lake Skärgölen.

Zn, µg/g dry w., perch liver

Bysjön
n(tot)= 160, n(year)= 16
mean=120 (120, 120)
slope= -12%(-2.8, -7.2)
CV(%)=3%, 1.7%, 7 yr
y(15)=107 (63.2, 703)
r=0.46, p=0.003

Stora Envättern
n(tot)= 160, n(year)= 16
mean=120 (120, 120)
slope= -12%(-2.8, -7.2)
CV(%)=3%, 1.7%, 7 yr
y(15)=107 (63.2, 703)
r=0.46, p=0.003

Skärgölen
n(tot)= 132, n(year)= 15
mean=116 (108, 124)
slope= -10%(-2.8, -7.2)
CV(%)=8%, 1.8%, 7 yr
y(15)=107 (63.2, 703)
r=0.48, p=0.002

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Figure 18.4. Zinc concentrations (µg/g dry weight) in perch liver from Lake Fiolen, Lake Hjärtsjön, and Lake Krageholmssjön.

Figure 18.5. Zinc concentrations (µg/g dry weight) in perch liver from Lake Remmarsjön, Lake Degervatnet, Lake Stensjön, and Lake Övre Skärsjön.
18.3 Summary

The highest concentration of zinc in perch liver was found in Lake Sännan in the County of Blekinge in 2013–2015. However, the results of the zinc concentrations are very homogenous among the lakes investigated within the monitoring programme.

Zinc is decreasing significantly at a majority of the perch sampling sites and in pike from Lake Storvindeln.
19 Arsenic – As

19.1 Introduction

19.1.1 Uses, Production and Sources
Arsenic is a natural component of the earth’s crust, and is found in all environmental media (World Health Organization, 2001). Major anthropogenic sources of environmental arsenic contamination are via industrial smelters, coal power plants and production and use of arsenic pesticides and herbicides (Eisler, 1994). An estimation of world arsenic production showed that copper chrome arsenate (CCA) used in timber treatment accounts for most arsenic use; however this source has recently decreased due to new arsenic compound regulations, which has seen the industry sector turn to arsenic-free preparations (KEMI, 2011a).

Elemental arsenic is produced by reduction of arsenic trioxide (As$_2$O$_3$) with charcoal. Arsenic trioxide is produced as a by-product of metal smelting operations, especially in copper smelting (World Health Organization, 2001, Eisler, 2007). Sweden was the world’s leading producer of arsenic trioxide, with ore from the Boliden area containing the highest levels of arsenic (Sveriges geologiska undersökning, 2005, Eisler, 2007).

19.1.2 Toxicological Effects
Acute, subacute and chronic effects can involve a number of organ systems including the respiratory, gastrointestinal, cardiovascular, nervous, and haematopoietic systems. Furthermore, disturbance of the liver function has been observed in both humans and animals after chronic exposure. There is also evidence that arsenic affects the heart in humans (United Nations Environment Programme et al., 1981).

In general, inorganic arsenic is more toxic than organic arsenic to aquatic biota, with trivalent, arsenic(III), species being more toxic than pentavalent arsenic(V). The toxic effects are modified by numerous biological factors such as water temperature, pH, organic content, phosphate concentration, suspended solids and the presence of other substances and toxicants (Eisler, 1994). Arsenic from water bioaccumulates in aquatic organisms, but there has been no evidence of biomagnification in the food web (Eisler, 1994, Sveriges geologiska undersökning, 2005).

19.1.3 Conventions, Aims and Restrictions

The European Union has, in a directive from the European Commission concerning restrictions on the marketing and use of arsenic compounds for the purpose of adapting Annex I to technical progress, states that arsenic compounds may not be used in the EU as substances and constituents of preparations intended for, amongst other things, the preservation of wood. Wood treated with arsenic compounds may not be placed on the EU market (The European Commision, 2006a).
19.1.4 Target Levels
No national target level for biota concerning arsenic is agreed upon.

Concentrations in water are usually < 10 μg/L (World Health Organization, 2001).

19.2 Results

19.2.1 Spatial Variation

![As, perch liver](image)

Figure 19.1. Spatial variation in concentration (μg/g dry weight) of arsenic in perch liver.

No general spatial pattern was observed for arsenic in perch liver. The highest concentrations of arsenic (2.79 and 1.88 μg/g dry weight respectively) were found in perch liver from Lake Remmarsjön in the County of Västernorrland and Lake Krageholmssjön in the County of Skåne in 2013-2015. The lowest concentration of arsenic (0.20 μg/g dry weight) came from perch liver from Lake Stora Skärsjön in the County of Västmanland (Fig. 19.1).

Arctic char from Lake Abiskojaure and pike from Lake Bolmen and Lake Storvindeln had similar concentrations of arsenic in 2012 (around 0.2 μg/g dry weight) (Fig. 19.2).

19.2.2 Temporal variation

Significant upward trends were observed for arsenic in Arctic char from Lake Abiskojaure as well as in pike from Lakes Storvindeln and Bolmen, 4.0, 5.7 and 3.0%, respectively (Fig. 19.2). Upward trends were observed for perch from Lakes Skärgölen, Krageholmssjön, and Remmarsjön, 7.6, 7.8, and 8.0% per year, respectively (Fig.19.3-19.5). No significant trends were seen for the other perch sites within the monitoring programme (Fig. 19.3–19.5).
The number of years required to detect an annual change of 10% for arsenic varies between 10–18 years for the pike, char and perch time series.

**Figure 19.2.** Arsenic concentrations (µg/g dry weight) in Arctic char liver (Lake Abiskojaure) and in pike liver (Lake Bolmen and Lake Storvindeln).
Figure 19.3. Arsenic concentrations (µg/g dry weight) in perch liver from Lake Bysjön, Lake Stora Envättern, and Lake Skärgölen.

Figure 19.4. Arsenic concentrations (µg/g dry weight) in perch liver from Lake Fiolen, Lake Hjärtsjön, and Lake Krageholmssjön.
Figure 19.5. Arsenic concentrations (µg/g dry weight) in perch liver from Lake Remmarsjön, Lake Degervattnet, Lake Stensjön, and Lake Övre Skärsjön.

19.3 Summary

The highest concentration of arsenic in perch liver was found in Lake Remmarsjön in the County of Västernorrland in 2013-2015.

No general temporal trend was observed for arsenic concentrations in fish liver. However, there are more significant upward trends and indications thereof than downward trends. This might be an indication that the arsenic concentration in freshwater biota in fact is increasing.
20 Silver – Ag

20.1 Introduction

20.1.1 Uses, Production and Sources
Silver is a noble metal (resistant to corrosion and oxidation) that occurs naturally, especially in sulfide-rich ores and in combination with other noble metals and copper, lead and zinc (Eisler, 1996, IVL, 2007). The main source of silver today is as a by-product in copper and lead smelting. In Sweden, silver is extracted in a copper mine near Gällivare, a lead mine at Arjeplog, and mines close to Skellefteå (IVL, 2007).

Anthropogenic sources of silver are mainly smelting operations, the manufacture and disposal of certain photographic and electrical supplies, coal combustion and cloud seeding (World Health Organization, 2002). Silver is used for jewellery, ornaments, tableware, utensils and currency (Eisler, 1996, World Health Organization, 2002, IVL, 2007). Electronics, batteries and solders containing silver present as solid waste may either be deposited in landfills or burnt in waste incinerators. Dispersal of residues in the environment may occur via leaching or emissions to the air (IVL, 2007).

Medicinally, silver is used for its bactericidal properties. Soluble silver compounds are used as antiseptic and bacteriostatic agents, as disinfectants (World Health Organization, 2002); and as antiseptic and antiodour agents in products such as in washing machines, refrigerators, socks and shoes (IVL, 2007). Metallic silver is used in amalgam dental fillings alloyed with mercury and small amounts of other metals (IVL, 2007).

Silver concentration in biota has been found to be higher near sewage outfalls, electroplating plants, mine waste and silver-iodide-seeded areas, than from more distant sites (Eisler, 1996).

20.1.2 Toxicological Effects
Silver has no known biological function in living organisms (IVL, 2007). It occurs naturally in several oxidation states. The most common states are elemental silver Ag⁰ and the monovalent cation Ag⁺. Soluble silver salts are generally more toxic than insoluble salts. As ionic Ag⁺, silver is one of the most toxic metals known to aquatic organisms in laboratory studies (Eisler, 1996, IVL, 2007). Silver has an affinity for suspended particles (Gill et al., 1994). In fish, silver has been found to induce the metal-binding protein metallothionein (IVL, 2007).

20.1.3 Conventions, Aims and Restrictions
Silver and all of the chemical compounds that emit silver or silver ions, should be regarded as a biocide product if its purpose is to prevent growth of bacteria. Silver used as a biocide product is restricted by the European directive 98/8/EC concerning the placing of biocidal products on the market (The European Parliament and The European Council, 1998).

20.1.4 Target Levels
No national target level for biota concerning silver is agreed upon.
The tolerable daily intake of silver for humans has been set at 5 µg/kg body weight (IRIS, 1991). WHO recommendations for the protection of groundwater, reports a critical concentration of 50 µg/L (World Health Organization, 2002).

Silver is comparably rare in the earth’s crust. The crustal abundance is estimated at 0.07 ppm (mg/kg), predominantly concentrated in basalt (Eisler, 1996). Average concentration of silver in natural waters is 0.2–0.3 µg/L (World Health Organization, 2002).

In Sweden, the analyses of background concentrations of silver have shown concentrations of 0.07 mg/kg in the fine particulate fraction of moraine, and 0.2 mg/kg in the fine fraction of sediment soils (Sveriges geologiska undersökning, 2005). In analysed lake sediments, measured concentrations were 0.16–0.66 mg/kg dry weight (Grahn et al., 2006), and 5–22 mg/kg dry weight (IVL, 2007). Background concentrations of silver in fish muscle from lakes have been measured as <0.21 µg/kg fresh weight (IVL, 2007).

20.2 Results

20.2.1 Spatial Variation

![Spatial Variation in Concentration of Silver in Perch Liver](image)

**Figure 20.1.** Spatial variation in concentration (µg/g dry weight) of silver in perch liver.
No general spatial pattern was observed for silver in perch liver. However, a few lakes had higher concentrations compared to the rest of the lakes. The highest concentration of silver (0.18 µg/g dry weight) was found in perch liver from Lake Remmarsjön in the County of Småland in 2013–2015. The lowest concentration of silver (0.0040 µg/g dry weight) came from perch liver from Lake Krageholmssjön in the County of Skåne (Fig. 20.1).

Pike from Lake Bolmen had slightly lower concentration compared to pike from Lake Storvindeln (0.12 and 0.16 µg/g dry weight respectively) while Arctic char from Lake Abiskojaure had almost twice as high concentration compared to pike from Lake Bolmen in 2013.

20.2.2 Temporal variation

A significant downward trend was observed for silver in perch liver from Lake Skärgölen at an annual mean decrease of 4.7% (Fig 20.3). In perch liver from Lake Stensjön and Lake Bysjön annual mean decreases of 8.1 and 8.0%, respectively, were seen for the last ten years in the times series (Fig 20.3, 20.5). A decrease in silver concentration was indicated in perch from Lake Fiolen and vice versa, an increase in silver concentration was indicated in perch from Lake Remmarsjön, during the last ten years at both sites (Fig 20.5). Whereas no significant trends were seen for the other matrices and sites within the monitoring programme (Fig. 20.2–20.5).

The number of years required to detect an annual change of 10% for silver varies between 10–21 years for the pike, char and perch time series.

Figure 20.2. Silver concentrations (µg/g dry weight) in Arctic char liver (Lake Abiskojaure) and in pike liver (Lake Bolmen and Lake Storvindeln).
Figure 20.3. Silver concentrations (µg/g dry weight) in perch liver from Lake Bysjön, Lake Stora Envättern, and Lake Skärölden.

Figure 20.4. Silver concentrations (µg/g dry weight) in perch liver from Lake Fiolen, Lake Hjärtsjön, and Lake Krageholmssjön.
Figure 20.5. Silver concentrations (µg/g dry weight) in perch liver from Lake Remmarsjön, Lake Degervatnet, Lake Stensjön, and Lake Övre Skärsjön.

20.3 Summary

The highest concentration of silver in perch liver was found in Lake Remmarsjön in the County of Västernorrland in 2013–2015.

The majority of the time series does not show any trend and it possible that the lakes with decreasing silver concentrations describe the local environment rather than any national trend. However, there are time series which shows significant decreasing concentrations but also a few where increasing concentrations are indicated during the last ten years.
21 Aluminium – Al

21.1 Introduction

21.1.1 Uses, Production and Sources
Aluminium is the most abundant metal in the Earth’s crust and is released from both natural and anthropogenic sources (Poléo, 1995). It has a silvery white to dull gray colour and natural occurring aluminium is present in the oxidation state 3+, i.e. aluminium is always combined with other elements in the nature, and form substances such as aluminosilicates, oxides, and hydroxides in rocks and minerals. Metallic aluminium is refined from bauxite ore which is known as the primary commercial source of aluminium. Oxides and silicates are two common forms of aluminium. Known isotopes of aluminium range from $^{21}$Al to $^{42}$Al; however, only $^{27}$Al occurs naturally.

As a metal, aluminium is a good thermal and electrical conductor. In addition, aluminium is widely used in aerospace industry, transportation, packaging, construction and household applications due to its low density, durability and corrosion resistance. Aluminium is also used as an emulsifying agent and an adjuvant in certain vaccines (Mitkus et al., 2011). Therefore, humans are exposed to aluminium via incidental ingestion of food and water containing aluminium or through vaccination. Aluminium can be used to form alloys with other metals and elements (copper, magnesium, silicon).

21.1.2 Environmental Fate
Atmospheric aluminium is derived from the weathering of aluminosilicate rocks and soils, coal combustion, cement manufacture, metal smelting and waste incineration (Jones and Bennett, 1986). In water, aluminium can form several complex ions depending on pH and the amount of dissolved organic matter (Priest, 2005). Dissolved aluminium which may be toxic to aquatic biota can only be found in water with pH < 5 (Jones and Bennett, 1986). Aluminium is present at a high abundance in soil and its chemical forms depend on the type and depth of the soil. Biologically available aluminium can be found in acidic soils, while insoluble forms may accumulate in the B horizon of podzolic soils (Jones and Bennett, 1986). Several plants can absorb aluminium from soils and accumulate very high concentration of aluminium.

21.1.3 Toxicological Effects
The health effects of aluminium are of interest especially because of its widespread occurrence in the environment and in commerce. Aluminium competes with calcium for absorption and bone mineralization. Therefore, increasing amounts of dietary aluminium may cause calcium deficiency and adverse effects on skeletal mineralization (Zafar et al., 2004). Aluminium is known as a neurotoxin though its role as an agent causing toxicity remains unclear (Banks and Kastin, 1989). In very high doses, aluminium can cause neurotoxicity, altering the function of the blood-brain barrier in rat (Banks and Kastin, 1989, Miu et al., 2003). Aluminium neurotoxicity may occur in humans since high concentration of aluminium was detected in elderly people with syndromes concerning impaired coordination and memory.
(Bowdler et al., 1979). Aluminium in some forms is acute toxic to fish and other aquatic organism, even though the level of aluminium is remarkably low in fresh water (Poléo, 1995). At pH from 5.0 to 6.0, Aluminium hydroxide bind and polymerize on the surface of fish gills, causing respiratory and ion regulatory dysfunctions.

21.1.4 Conventions, Aims and Restrictions
The World Health Organization on esthetic grounds specifies that aluminium concentrations in tap water should not exceed 200 µg/L (Priest, 2005).

21.1.5 Target Levels
No national target level for biota concerning aluminium is agreed upon.

21.2 Results

21.2.1 Spatial Variation

![Al, perch liver](image)

**Figure 21.1.** Spatial variation in concentration (µg/g dry weight) of aluminium in perch liver.

No general spatial trend for aluminium in perch liver is observed. The highest concentration of aluminium in perch liver were found in Lake Sännen (50µg/g dry weight) in the County of Blekinge in 2013–2015. The lowest concentration of aluminium (2.5 µg/g dry weight) came from perch liver from Lake Krageholmsjön in the County of Skåne (Fig. 21.1).

The concentration of aluminium in pike is similar at the two lakes (about 1.9 µg/g dry weight) in 2012, while the concentration in Arctic char is about twice as high (4.0 µg/g dry weight) (Fig. 21.2).
21.2.2 Temporal variation

A significant downward trend was seen in perch from Lake Övre Skärsjön, annual mean decrease of 3.5% during 2000–2015 (Fig. 21.5), and in Lake Fiolen with an annual mean decrease of 6.6%, for the last ten years (Fig. 21.4). Furthermore, a decrease in aluminium concentration was indicated in perch from Lake Skärgölen during the whole monitoring period. No significant trends were seen for aluminium for the other matrices and sites within the monitoring programme (Fig. 21.2–21.5).

The number of years required to detect an annual change of 10% for aluminium varies between 10–15 years for the pike, char and perch time series.

![Figure 21.2. Aluminium concentrations (µg/g dry weight) in Arctic char liver (Lake Abiskojaure) and in pike liver (Lake Bolmen and Lake Storvindeln).](image-url)
Figure 21.3. Aluminium concentrations (µg/g dry weight) in perch liver from Lake Bysjön, Lake Stora Envättern, and Lake Skärgölen.

Figure 21.4. Aluminium concentrations (µg/g dry weight) in perch liver from Lake Fiolen, Lake Hjärtsjön, and Lake Krageholmssjön.
Figure 21.5. Aluminium concentrations (μg/g dry weight) in perch liver from Lake Remmarsjön, Lake Degervattnet, Lake Stensjön, and Lake Övre Skärsjön.

21.3 Summary

The highest concentrations of aluminium in perch liver were found in Lake Sännen in the County of Blekinge in 2013-2015.

No general temporal trend was observed for aluminium concentrations in fish liver.
22 Bismuth – Bi

22.1 Introduction

22.1.1 Uses, Production and Sources
Bismuth is a relatively rare element since its abundance in the Earth's crust is estimated to be about 0.2 ppm (Jayasinghe et al., 2004, Das et al., 2006). Bismuth is seldom found in its elemental state (as a pure metal) in the earth. Its compounds are generally found along with ores of other metals, such as lead, silver, gold, and cobalt. The most important mineral of bismuth are bismuthinite, also known as bismuth glance (Bi₂S₃) and bismite (Bi₂O₃) (Hammond, 2004, Jayasinghe et al., 2004). Bismuth is mainly produced as a by-product from lead and copper smelting.
There is only one naturally occurring isotope of bismuth, ²⁰⁹Bi. Bismuth is used in semiconductors, cosmetic products, alloys, catalysts in the chemistry industry, metallurgical additives, and preparation and recycling of uranium in nuclear fuels (Das et al., 2006). Alloys of bismuth can be made into pellets and used as an alternative for lead shot (Jayasinghe et al., 2004). In medicine, it is used in the treatment of gastrointestinal tract disturbance, e.g. gastritis and peptic ulcer (Gorbach, 1990).

22.1.2 Toxicological Effects
The Food and Agricultural Organization and the World Health Organization do not classify bismuth as an essential element for the body (Das et al., 2006). Bismuth compounds will be excreted through urine in case it is absorbed. Because of this, bismuth is considered to be of low toxicity. However, a number of toxic effects have been found in humans and other animals at certain concentrations. The use of bismuth-containing medication was reduced as a result of reports concerning severe neurological symptoms as well as liver and kidney damages associated with intake of bismuth salts (Ross et al., 1996, Burguera et al., 1999). Bismuth can enter the nervous system of mice and affect motor neuron (Das et al., 2006).
There is limited information on the effects and environmental fate of bismuth, but in general, bismuth is considered to have a small environmental impact and is less toxic than its other neighbours in the periodic table (lead, antimony, and polonium).

22.1.3 Target Levels
No national target level for biota concerning bismuth is agreed upon.

22.2 Results
No new data has been collected since 2012 for bismuth, but the results from previous years are included for the benefit of the reader.
22.2.1 Spatial Variation

No general spatial trend for bismuth in perch liver is observed. The highest concentration of bismuth (0.03 µg/g dry weight) was found in perch liver from Lake Sännern in the County of Blekinge in 2010–2012. The lowest concentration of bismuth (0.002 µg/g dry weight) in perch liver is reported for Lake Krankesjön in the County of Skåne and Lake Algjuttern in Kalmar County (Fig. 22.1).

Pike from Lake Storvindeln and Lake Bolmen had the same magnitude of concentration of bismuth (0.002 µg/g dry weight) and the concentration of bismuth in Arctic char from Lake Abiskojaure was half of the amount, 0.001 µg/g dry weight (Fig. 22.2).

22.2.2 Temporal variation

In perch from Lake Bysjön a significant upward trend was seen for bismuth with an annual mean increase of 8.5%. A significant downward trend was observed for bismuth in perch from Lake Fiolen of 5.3% for the whole monitoring period, as well as in perch from Lake Hjärtsjön with a mean decrease of 8.2% per year the last ten years. No significant trends were seen for bismuth for the other matrices and sites within the monitoring programme (Fig.22.2–22.5).

The number of years required to detect an annual change of 10% for bismuth varies between 7–15 years for the pike, char and perch time series.
Figure 22.2. Bismuth concentrations (µg/g dry weight) in Arctic char liver (Lake Abiskojaure) and in pike liver (Lake Bolmen and Lake Storvindeln).

Figure 22.3. Bismuth concentrations (µg/g dry weight) in perch liver from Lake Bysjön, Lake Stora Envättern, and Lake Skärölen.
Figure 22.4. Bismuth concentrations (µg/g dry weight) in perch liver from Lake Fiolen, Lake Hjärtsjön, and Lake Krageholmsjön.

Figure 22.5. Bismuth concentrations (µg/g dry weight) in perch liver from Lake Remmarsjön, Lake Degervattnet, Lake Stensjön, and Lake Övre Skarsjön.
22.3 Summary

The highest concentration of bismuth in perch liver (2010-2012) was found in Lake Sännen in the County of Blekinge.

No general temporal trend was observed for bismuth concentrations in fish liver.
23 Tin – Sn

23.1 Introduction

23.1.1 Uses, Production and Sources

Tin is a chemical element with chemical similarities to both lead and germanium. It has two oxidation states, stannous Sn\(^{+2}\) and stannic Sn\(^{+4}\), the latter is slightly more stable (Goyer R.A. and Clarkson T.W., 2001). Tin is mainly obtained from the mineral cassiterite and has the greatest number of stable isotopes, \(^{10}\text{Sn}, ^{114}\text{Sn}, ^{115}\text{Sn}, ^{116}\text{Sn}, ^{117}\text{Sn}, ^{118}\text{Sn}, ^{119}\text{Sn}, ^{120}\text{Sn}, ^{122}\text{Sn}, ^{124}\text{Sn}, ^{126}\text{Sn}, \) and where three of them are potentially radioactive. There are also 29 additional unstable isotopes.

The most commercially significant inorganic tin compounds include tin(II) chloride, tin(IV) chloride, tin(IV) oxide, potassium and sodium stannates (SnO\(_3\)^{2-}), tin(II) fluoride, tin(II) difluoroborate, and tin(II) pyrophosphate (World Health Organization, 2005). Of these, tin(II) chloride is the most important compound, and it is predominantly used as a reducing agent in organic and inorganic synthesis and in manufacture of metallized glazing, glass, and pigments. Moreover, tin(IV) chloride is also an important compound and it is used in organic synthesis, in plastics, as an intermediate in organotin compound manufacture, and in the production of tin(IV) oxide films on glass (World Health Organization, 2005).

Tin does not oxidize easily in air, and therefore it is used for coating other metals in order to prevent corrosion. Tin is thus used in many alloys, particularly in soft solders, which accounts for about 34% of all tin produced. It is also largely used as corrosion-resistant tin-plating of steal. Another 25–30% of the produced tin is used as a protective coating for other metals, especially food containers, because of its low toxicity. Organic tin compounds have been used in fungicides, bactericides, and slimicides, and it has also been used as stabilizer in plastics (Goyer R.A. and Clarkson T.W., 2001).

23.1.2 Environmental Fate

Tin can be released to the environment both by natural and anthropogenic sources. Tin occurs naturally in the Earth’s crust. Both organic and inorganic forms of tin compounds are found in various environmental media. Tin is a component in many soils, and inorganic tin compounds may be released in dusts from wind storms, roads, and agricultural activities. Tin can be released from production, use, disposal, and recovery of tin and tin compounds. Gases, dusts, and fumes that contain tin may be released from smelting and refining processes, industrial uses of tin, waste incineration, and burning of fossil fuels. Tin may also be released to soil from landfills of tin-containing wastes, including used cans. Moreover, the degradation of organotin can result in the formation of inorganic tin.

23.1.3 Toxicological Effects

Inorganic tin salts have low toxicity. They are poorly absorbed and rapidly excreted (Winship, 1988). However, some organic tin compounds are highly toxic. The two most toxic known
organotins are trimethyltin and triethyltin (Winship, 1988, Goyer R.A. and Clarkson T.W., 2001).

Inorganic tin compounds have low toxicity in both aquatic and terrestrial organisms. This is mainly due to their low solubility, poor absorption, low accumulation in tissues, and rapid excretion.

Toxicity tests on fish show that tin(II) chloride is more toxic than tin(IV) chloride, 96h LC₅₀ test for fish ranges from 35 mg of tin(II) per litre to > 1000 mg tin(IV) per litre. The 7- and 28-day LC₅₀ embryo-larval test for fish ranged from 0.4 to 2.1 mg/Litre for tin(II) (World Health Organization, 2005).

The main human exposure to tin is through food. Those that routinely consume canned fruit, vegetables, and juices from unlacquered cans could ingest 50–60 mg tin daily (World Health Organization, 2005).

23.1.4 Target Levels

No national target level for biota concerning tin is agreed upon.
23.2 Results

23.2.1 Spatial variation

Almost all values for tin in perch liver were in 2013–2015 below LOQ, regardless of lake so therefore a map is not presented. The value for LOQ ranges between 0.048–0.30µg/g dw.

The concentration of tin in pike liver from Lake Bolmen and Lake Storvindeln and in Arctic char liver from Lake Abiskojaure ranges between 0.05–0.06 µg/g dry weight (Fig. 23.1).

23.2.2 Temporal variation

Significant upward trends for perch were observed in Lake Skärgölen and Lake Remmarsjön from 2000/2001–2015 with an annual increase of 7.1 and 16%, respectively (Fig. 23.2, 23.4). Furthermore, in Arctic char from Lake Abiskojaure and pike from Lakes Bysjön, Fiolen, Krageholmssjön and Stensjön there are downward trends during the last ten years of monitoring (Fig. 23.1–23.4). There are also indications of downward trends in pike from Lake Storvindeln and in perch from Lakes Skärgölen, Hjärtsjön and Degervattnet during the most recent ten years.

The number of years required to detect an annual change of 10% for tin varies between 14–21 years for the pike, char and perch time series.
Figure 23.1. Tin concentrations (µg/g dry weight) in Arctic char liver (Lake Abiskojaure) and in pike liver (Lake Bolmen and Lake Storvindeln).

Figure 23.2. Tin concentrations (µg/g dry weight) in perch liver from Lake Bysjön, Lake Stora Envättern, and Lake Skärgölen.
Figure 23.3. Tin concentrations (µg/g dry weight) in perch liver from Lake Fiolen, Lake Hjärtsjön, and Lake Krageholmssjön.

Figure 23.4. Tin concentrations (µg/g dry weight) in perch liver from Lake Remmarsjön, Lake Degervattnet, Lake Stensjön, and Lake Övre Skärsjön.
23.3 Summary

Almost all values of tin concentrations were below LOQ in 2013–2015.

Downward trends were seen in Arctic char during the whole time period and in perch during the most recent ten years at several lakes. But in perch from Lake Skärgölen and Remmarsjön significant upward trends in tin concentrations were observed. Pike did not show any trends in tin concentration.
24 PCBs, Polychlorinated biphenyls

24.1 Introduction

24.1.1 Usage, Production and Sources
Polychlorinated biphenyls (PCBs) consist of two linked phenyl rings substituted with one or more chlorine atoms. Out of a possible 209 congeners, depending on the number and position of the chlorine atoms, 20 have non-ortho chlorine substitutions and can thus attain a planar structure. Similar to the highly toxic polychlorinated dibenzo-p-dioxins and dibenzofurans (Mckinney et al., 1985) Serico et al. 1991), see chapter 29. PCBs are synthetic chemicals that have been used in a wide variety of manufacturing processes, especially as plasticizers, insulators and fire retardants. PCBs are widely distributed in the environment through inappropriate handling of waste material or e.g., leakage from large capacitors and hydraulic systems.

24.1.2 Toxicological Effects
PCBs can influence human health by affecting multiple organ systems (Carpenter, 1998, ATSDR, 2000, Carpenter, 2006) and their toxicological effects on e.g., reproduction in mink, is well documented (Aulerich and Ringer, 1977, Jensen S. et al., 1977, Bleavins et al., 1980).

24.1.3 Conventions, Aims and Restrictions
In 1992, HELCOM revised the PCBs for which special bans and restrictions on transport, trade, handling, use and disposal were imposed. The Minister Declaration from 1988, within HELCOM, calls for a reduction of stable organic substances by 50% by 1995, with 1987 as the base year (HELCOM, 1988).

The Minister Declaration from 1996, within HELCOM, and the declaration in Esbjerg 1995, calls for measures for toxic, persistent, bioaccumulating substances like PCBs to have ceased completely in the year 2020 (The Esbjerg declaration, 1995).

PCBs are one of the initial 12 Persistent Organic Pollutants (POPs) included in The Stockholm Convention on POPs, an international agreement requiring measures for reducing or preventing release of dangerous substances into the environment (The Stockholm Convention, 2008).

In 1973, PCB use was banned in Sweden (SFS 1971:385), except for within sealed systems. In 1978, all new use of PCBs was forbidden followed by a total ban of PCBs in Sweden in 1995 (SFS 1995: 1095).

24.1.4 Target Levels
The target levels used for CB-153 and CB-118 in the time series for fish are 1600 and 24 µg/kg lipid weight respectively. For further information on target levels and selection of target level see chapter 10.
24.2 Results

24.2.1 Spatial Variation

Figure 24.1. Spatial variation in concentration (µg/g lipid weight) of CB-118 in perch muscle.
In recent years, PCBs have only been analysed in perch from six lakes within the national Swedish monitoring programme for contaminants in freshwater biota. However, in 2013 all lakes within the monitoring programme were analysed for CB-118 and CB-153. But this means that in the maps most lakes only show a value from 2013 while a mean value for 2013-2015 shows for the lakes where annual results are available. The highest concentration of CB-118 (0.09 µg/g lw) and CB-153 (0.48 µg/g lw) in perch muscle was found in Lake Fräcksjön in the County of Västra Götaland in 2013. The lowest concentration of CB-118 and CB-153 (0.004 and 0.015 µg/g lw respectively) in perch muscle was determined in Lake Remmarsjön in the County of Västernorrland in 2013 (Fig. 24.1–24.2).

The concentration of CB-153 and CB-118 in pike from Lake Bolmen is about three times higher than in pike from Lake Storvodeln and about ten times higher than in perch and Arctic char from the lakes monitored within the programme (Fig. 24.3–24.6).

**24.2.2 Temporal variation**

The pike and char time series show significant downward trends for CB-118 of about 4.0–6.7% per year in the lakes that are monitored (Fig. 24.3). No trend is detected for CB-118 in the perch time series from Lake Skärgölen and Lake Stensjön (Fig. 24.4).

CB-153 shows similar decreasing trends for pike and char as CB-118 (with the exception of pike from Lake Bolmen where a trend is only indicated), with an annual decrease of about 3.1–6.4% (Fig. 24.5). No trend for CB-153 is seen in perch from Lake Skärgölen or Lake Stensjön (Fig. 24.6).

The trends for CB-153 and CB-118 are consistent with and of similar magnitude to other time series for PCBs in marine biota (Bignert A. et al., 2010).
The number of years required to detect an annual change of 10% for CB-118 varies between 9–16 years for the pike, char and perch time series.

The number of years required to detect an annual change of 10% for CB-153 varies between 8–17 years for the pike, char and perch time series.

Figure 24.3. CB-118 concentrations (µg/g lipid weight) in Arctic char muscle (Lake Abiskojaure and Lake Tjultråsk) and in pike muscle (Lake Bolmen and Lake Storvindeln). The green area denotes the levels below the suggested target value for CB-118 in fish.
Figure 24.4. CB-118 concentrations (µg/g lipid weight) in perch muscle (Lake Skärgölen and Lake Stensjön). The green area denotes the levels below the suggested target value for CB-118 in fish.

Figure 24.5. CB-153 concentrations (µg/g lipid weight) in Arctic char muscle (Lake Abiskojaure and Lake Tjulträsk) and in pike muscle (Lake Bölmén and Lake Storvindeln). The green area denotes the levels below the suggested target value for CB-153 in fish.
Figure 24.6. CB-153 concentrations (µg/g lipid weight) in perch muscle (Lake Skärgölen and Lake Stensjön). The green area denotes the levels below the suggested target value for CB-153 in fish.
24.2.3 Comparison to thresholds

In all areas and species, CB-153 concentration is below the suggested target level based on the OSPAR EAC (Environmental Assessment Criteria) of 1.6 µg/g lipid weight (OSPAR Commision, 2009).

The suggested target level for CB-118 based on the OSPAR EAC of 0.024 µg/g lipid weight were exceeded in perch from the Lakes Krankesjön, Fysingen, Fräcksjön, Sännem, Tärnan, and Stora Skärjsjön (Fig. 24.7) and in pike from Lake Bolmen (OSPAR Commision, 2009).

Figure 24.7. Spatial variation in concentration (µg/g lipid weight) of CB-118 in perch muscle. The green sections of the bars are representing concentrations under the threshold level (0.024 µg/g lipid weight) and the red sections concentrations above.

24.3 Summary

The highest concentration in perch of CB-118 and CB-153 was found in Lake Fräcksjön in the County of Västra Götaland in the most recent three years.

CB-118 and CB-153 concentrations varied between species and sites; however temporally, the concentration has decreased by approximately 3–7% per year (with a few exceptions) in the freshwater environment since the end of the 1960/70s.

In all areas and species, CB-153 concentration is below the suggested target level, whereas the target level for CB-118 is exceeded in perch from the Lakes Krankesjön, Fysingen, Fräcksjön, Sännem, Tärnan, and Stora Skärjsjön, and in pike from Lake Bolmen.
25 DDTs, Dichlorodiphenylethanes

Updated 16.10.31

25.1 Introduction

25.1.1 Usage, Production and Sources

DDT is a persistent synthetic pesticide that primarily degrades to DDE and DDD. It is still used in some countries as it is an effective pesticide against mosquitoes and hence malaria. The presence of DDT and its metabolites in the Arctic area indicates long range transport (Welch et al., 1991).

25.1.2 Toxicological Effects

DDT has severe health effects on wildlife. In fish-eating birds, reduced reproductive success is widely documented e.g., reduced productivity in top predator white-tailed sea eagles (Helander et al., 2008). This is due to several factors such as failure to return to nesting sites, egg shell thinning, inability of eggs to hatch, reduced number of reproducing pairs and nestling brood size (Helander et al., 2008, Hamlin H.J. and Guillette Jr L.J., 2010). Also, embryo mortality, thyroid malfunction, and immunosupression have been documented (Hamlin H.J. and Guillette Jr L.J., 2010).

25.1.3 Conventions, Aims and Restrictions

In 1992, the Helsinki Convention (HELCOM) revised the DDTs for which special bans and restrictions on transport, trade, handling, use and disposal were imposed. The Minister Declaration from 1988, within HELCOM, calls for a reduction of stable organic substances by 50% by 1995, with 1987 as the base year (HELCOM, 1988).

DDT is one of the initial 12 Persistent Organic Pollutants (POPs) included in The Stockholm Convention on POPs, an international agreement requiring measures for reducing or preventing release of dangerous substances into the environment.

The Stockholm Convention was adopted in 2001 and entered into force in 2004 (Stockholm Convention).

In Sweden, DDT was partially banned as a pesticide in 1970, and completely banned in 1975 due to its persistence and environmental impact.

In western European countries, the use of DDT ceased around 1990, although heavy use was banned between 1970 and 1975 in most countries bordering the Baltic Sea.

25.1.4 Target Levels

The target level used for DDE in the time series for fish is 5 µg/kg wet weight. For further information on target levels and selection of target level see chapter 10. The original target level has been recalculated for each time series based on the lipid percentage. The
recomputed target level (Tv) together with the lipid percentage (lp) is shown above the statistical information in each time series.

25.2 Results

25.2.1 Spatial Variation

![Diagram showing spatial variation in concentration (µg/g lipid weight) of DDE in perch muscle.](#)

**Figure 25.1.** Spatial variation in concentration (µg/g lipid weight) of DDE in perch muscle.
Figure 25.2a. Spatial variation in concentration (µg/g lipid weight) of DDT in perch muscle.

Figure 25.2b. Spatial variation in concentration (µg/g lipid weight) of DDT in perch muscle, with Lake Fräcksjön excluded.
In recent years DDTs are only analysed in perch from six lakes within the national Swedish monitoring programme for contaminants in freshwater biota. However, in 2013 all lakes within the monitoring programme were analysed for DDTs. Thus, in the maps most lakes only show a value from 2013 while a mean value for 2013-2015 is shown for the lakes where annual results are available. The highest concentration of DDE (0.33 µg/g lw) in perch muscle was found in Lake Fräcksjön in the County of Västra Götaland in 2013 and the lowest concentration (0.021 µg/g lw) was found in Lake Stor-Backsjön in the County of Jämtland (Fig. 25.1). In 2013, one extremely high value of DDT was found in Lake Fräcksjön (0.24 µg/g lw) in the County of Västra Götaland (Fig. 25.2a). In Fig. 25.2b Lake Fräcksjön has been removed in order to get a clearer picture of the spatial variation of the other lakes. Most other lakes are in the range 0.005 to 0.008 µg/g lw. The lowest concentration of DDT (0.0054 µg/g lw) was found in Lake Spjutsjön in the County of Dalarna (Fig. 25.2).

The concentration of DDE in pike from Lake Bolmen is about seven times higher than in pike from Lake Storvindeln and almost 20 times higher than in Arctic char and between three and six times higher than in perch from the time series lakes monitored within the programme (Fig. 25.3–25.6).

25.2.2 Temporal variation

Temporal trends of DDE and DDT in char from Lake Abiskojaure and Lake Tjulträsk and in pike from Lake Bolmen and Lake Storvindeln show similar patterns (Fig. 25.3, 25.5). The pesticide has decreased significantly during the period 1967/68/81–2015, with an annual mean decrease of 5.8–8.9 and 4.3–11% for DDE and DDT, respectively.

DDE and DDT show significant downward trends in perch from Lake Skärgölen with an annual change of 4.9 and 6.1 %, respectively, during the period 1980–2015 (Fig. 25.4, 25.6). In Lake Stensjön, there is a significant downward trend of DDT concentration in perch muscle with an annual mean decrease of 5.1%. No trend is detected for DDE in perch from Lake Stensjön (Fig. 25.4, 25.6), but the lake has only been monitored for 15 years, with a gap between 2000 and 2005. This time period is too short to be able to detect a trend of 5–10% annual change with this material.

The concentration of DDT is close to or at the LOQ (level of quantification) for most of the matrices and sites that are monitored during recent years.

The number of years required to detect an annual change of 10% for DDE varies between 12–17 years for the pike, char and perch time series.

The number of years required to detect an annual change of 10% for DDT varies between 14–20 years for the pike, char and perch time series.
Figure 25.3. DDE concentrations (µg/g lipid weight) in Arctic char muscle (Lake Abiskojaure and Lake Tjulträsk) and in pike muscle (Lake Bolmen and Lake Storvindeln). The green area denotes the levels below the suggested target value for DDE in fish.

Figure 25.4. DDE concentrations (µg/g lipid weight) in perch muscle (Lake Skärgölen and Lake Stensjön). The green area denotes the levels below the suggested target value for DDE in fish.
Figure 25.5. DDT concentrations (µg/g lipid weight) in Arctic char muscle (Lake Abiskojaure and Lake Tjulträsk) and in pike muscle (Lake Bolmen and Lake Storvindeln).

Figure 25.6. DDT concentrations (µg/g lipid weight) in perch muscle (Lake Skårgölen and Lake Stensjön).
25.2.3 Comparison to thresholds
In all areas and species, DDE concentration is below the suggested target level based on the OSPAR EAC (Environmental Assessment Criteria) of 0.005 µg/g wet weight.

25.3 Summary

The highest concentration of DDE and DDT in perch muscle was found in Lake Fräcksjön in the County of Västra Götaland in the most recent three years.

DDE and DDT concentrations varied between species and sites; however temporally, the concentration has decreased by approximately 5–20% per year in the freshwater environment since the end of the 1970s.

In all areas and species, DDE concentration is below the suggested target level.
26 HCHs, Hexachlorocyclohexanes

The isomers α-HCH, β-HCH and γ-HCH i.e., lindane, have been analysed in muscle tissue for various fish species since 1980. The concentrations of β-HCH are in many cases close to the quantification limit, which implies analytical problems.

26.1 Introduction

26.1.1 Uses, Production and Sources
Technical HCH contains various isomers: 60–75% α-HCH; 15% γ-HCH (lindane); 7–10% β-HCH; 7% δ-HCH; and 1–2% ε-HCH, and came into general use in 1950 (Gaul H., 1992). The γ-isomer is the most toxic isomer of the HCHs, being 500–1000 times as potent as the α-isomer (White-Stevens R., 1971).

26.1.2 Conventions, Aims and Restrictions
The Minister Declaration from 1988, within HELCOM, calls for a reduction of stable organic substances by 50% by 1995, with 1987 as the base year (HELCOM, 1988).

HCHs, α-HCH, β-HCH and γ-HCH, is included as Persistent Organic Pollutants (POPs) in The Stockholm Convention on POPs, an international agreement requiring measures for reducing or preventing release of dangerous substances into the environment (Stockholm Convention).

In Sweden, the use of lindane was severely restricted in 1970, and in 1978 prohibited for use in agriculture due to suspected carcinogenic properties and persistence. Remaining use was banned in 1988/89.

The use of technical HCH stopped in countries around the Baltic between 1970 and 1980. Since 1980, use of lindane in Europe has been allowed only as an insecticide. In a study investigating global HCH usage it found that both technical HCH and technical Lindane was used to a great extent in France and Italy as recently as 1990 (Li et al., 1996).

26.1.3 Target Levels
The target level used for ΣHCH in the time series for fish is 26 µg/kg wet weight. For further information on target levels and selection of target level see chapter 10. The original target level has been recalculated for each time series based on the lipid percentage. The recalculated target level (Tv) together with the lipid percentage (lp) is shown above the statistical information in each time series.

26.2 Results
26.2.1 Spatial Variation

In recent years HCHs are only analysed in perch from six lakes within the national Swedish monitoring programme for contaminants in freshwater biota. However, in 2013 all lakes within the monitoring programme were analysed for HCHs. Thus, in the maps most lakes only show a value from 2013 while a mean value for 2013–2015 is shown for the lakes where annual results are available. The concentrations are rather homogenous among the lakes, but the highest concentration of $\alpha$-HCH, (0.0057 µg/g lw) was found in perch from Lake Sännen in the County of Blekinge in 2013 (Fig. 26.1). The levels in the other lakes that are analysed for $\alpha$-HCH are close to or at LOQ (Fig. 26.1). The concentrations of $\beta$-HCH and lindane in perch muscle were almost all under LOQ in 2013–2015 and are therefore not presented as maps.

The concentration of lindane is of similar magnitude in all of the investigated lakes (Fig. 26.2–26.3).

26.2.2 Temporal variation

Temporal trends of lindane in Arctic char from Lake Abiskojaure and Lake Tjulträsk, and in pike from Lake Bolmen and Lake Storvindeln, show similar patterns (Fig. 26.2). Lindane has decreased significantly during the period 1980–2015, with an annual change of 4.2–9.8%. During the last ten years, levels of lindane are close to or at LOQ.
Lindane shows significantly downward trends in perch from Lake Stensjön and Lake Skärgölen during the whole time period, with an annual mean change of 3.4 and 2.1%, respectively (Fig. 26.3).

∑HCH has decreased significantly in Arctic char from Lake Abiskojaure and in pike from Lake Bolmen and Lake Storvindeln with an annual mean decrease of 5.1, 3.8, and 1.7%, respectively.

∑HCH has decreased significantly in perch from Lake Stensjön and Lake Skärgölen during the whole time period with an annual mean change of 1.8 and 1.6% respectively (Fig. 26.5).

The number of years required to detect an annual change of 10% for lindane varies between 7–18 years for the pike, char and perch time series.

The number of years required to detect an annual change of 10% for ∑HCH varies between 6–14 years for the pike, char and perch time series.

Figure 26.2. Lindane concentrations (µg/g lipid weight) in Arctic char muscle (Lake Abiskojaure and Lake Tjulträsk) and in pike muscle (Lake Bolmen and Lake Storvindeln).
Figure 26.3. Lindane concentrations (µg/g lipid weight) in perch muscle (Lake Skärgölen and Lake Stensjön).

Figure 26.4. ∑HCH concentrations (µg/g lipid weight) in Arctic char muscle (Lake Abiskojaure and Lake Tjulträsk) and in pike muscle (Lake Bolmen and Lake Storvindeln). The green area denotes the levels below the suggested target value for ∑HCH in fish.
26.2.3 Comparison to thresholds

In all areas and species, $\Sigma$HCH concentration is below the suggested target level based on The Swedish Environmental Research Institute’s (IVL) conversion of the EQS for surface water to biota of 0.026 µg/g wet weight (Fig. 26.4–26.5).

26.3 Summary

The highest concentration of $\alpha$-HCH in perch muscle was found in Lake Sännin in the County of Blekinge in the most recent three years.

The lindane concentrations varied between species and sites; however temporally, the concentration has decreased by approximately 2.1–9.8% per year in the freshwater environment since the end of the 1960/70s.

In all areas and species, $\Sigma$HCH concentration is below the suggested target level.
27 HCB, Hexachlorobenzene

27.1 Introduction

27.1.1 Uses, Production and Sources
The use of the highly persistent HCB as a fungicide is banned in the Baltic countries. Although it may still reach the environment as a by-product of many chlorinating processes, for example pentachlorophenol and vinyl chloride monomer production, we have reason to expect a decrease in biological samples from the Baltic.

27.1.2 Conventions, Aims and Restrictions
The Minister Declaration from 1988, within HELCOM, calls for a reduction of stable organic substances by 50% by 1995, with 1987 as the base year (HELCOM, 1988).

HCB is one of the initial 12 Persistent Organic Pollutants (POPs) included in The Stockholm Convention on POPs, an international agreement requiring measures for reducing or preventing release of dangerous substances to the environment (Stockholm Convention).

In 1980, HCB was withdrawn from the market in Sweden because of its carcinogenic effects on experimental animals and its persistence.

The use of HCB as a fungicide has been banned in the Baltic countries since mid-1970s (Gaul 1992).

27.1.3 Target Levels
The target level used for HCB in the time series for fish is 10 µg/kg wet weight. For further information on target levels and selection of target level see chapter 10. The original target level has been recalculated for each time series based on the lipid percentage. The recalculated target level (Tv) together with the lipid percentage (lp) is shown above the statistical information in each time series.

27.2 Results

27.2.1 Spatial Variation
The concentrations of HCB in perch muscle were almost all under the LOQ in 2013–2015 and are therefore not presented in a map.

The concentrations of HCB in pike and Arctic char (Lake Bolmen, Lake Storvindeln, Lake Tjulträsk, and Lake Abiskojaure) are of similar magnitude (0.01 µg/g lipid weight), but with about twice the concentrations found in perch from the lakes monitored within the programme (Fig. 27.1–27.2).
27.2.2 Temporal variation

Temporal trends of HCB in char from Lake Abiskojaure and in perch from Lake Skärgölen show decreasing concentrations of HCB, mean decrease of 2.4 and 1.8%, respectively (Fig. 27.1, 27.2). No such trend is detected in char from Lake Tjulträsk, pike from Lake Bolmen and Lake Storvindeln, or in perch from Lake Stensjön (Fig. 27.1, 27.2). Instead a significant increase during the last ten years is seen in pike from Lake Bolmen and Lake Storvindeln, with an annual mean increase of 5.6 and 6.8%, respectively (Fig. 27.1).

The number of years required to detect an annual change of 10% for HCB varies between 5–17 years for the pike, char and perch time series.

Figure 27.1. HCB concentrations (µg/g lipid weight) in Arctic char muscle (Lake Abiskojaure and Lake Tjulträsk) and in pike muscle (Lake Bolmen and Lake Storvindeln). The green area denotes the levels below the suggested target value for HCB in fish.
27.2.3 Comparison to thresholds

In all areas and species, HCB concentration is below the suggested target level based on the EQS\textsubscript{biota} of 0.010 µg/g wet weight (Fig. 27.1 and 27.2).

27.3 Summary

The HCB concentrations varied in some cases between species and sites. The concentration has decreased by approximately 2% per year in one of the char time series (Lake Abiskojaure) since the end of the 1970s and at the same magnitude for perch (Lake Skärgölen) since the end of 1990s. However, two of the pike time series shows significant increase of HCB concentrations during the last ten years.

In all areas and species, HCB concentration is below the suggested target level.
28 PFASs, Perfluoroalkyl substances

28.1 Introduction

The different PFASs monitored in the freshwater programme include perfluorohexanoic acid (PFHxA), perfluoroheptanoic acid (PFHpA), perfluorooctanoic acid (PFOA), perfluorononanoic acid (PFNA), perfluorodecanoic acid (PFDA), perfluoroundecanoic acid (PFUnDA), perfluorododecanoic acid (PFDoDA), perfluorotridecanoic acid (PFTrDA), perfluorotetradecanoic acid (PFTeDA), perfluoropentadecanoic acid (PFPeDA), perfluorobutanesulfonic acid (PFBS), perfluorohexanesulfonic acid (PFHxS), perfluorooctanesulfonic acid (PFOS), perfluorodecanesulfonic acid (PFDS) and perfluorooctane sulfonamide (FOSA).

28.1.1 Uses, Production and Sources

Perfluoroalkyl substances (PFASs) are anthropogenic surfactants with exceptional stability and surface tension lowering potential. PFASs have been used industrially (e.g., production of fluoropolymers) and commercially (water and stain proofing agents and fire-fighting foams) since the beginning of the 1950s. It was not until the year 2000 that the main producer, 3M, started to phase out production of the main compounds of concern, perfluorooctane sulfonate (PFOS) and PFOS derivatives, perfluorooctanoate (PFOA) as well as perfluorohexane sulfonate (PFHxS) (Buck et al., 2011).

Environmental PFAS contamination has multiple emission sources. These include primary emissions of PFASs to air and water from industrial production and application, as well as secondary emissions from consumer products or via sewage treatment plant effluents. For the persistent perfluoroalkyl acids (PFAAs) a further distinction can be made between direct sources from manufacturing and use of PFAAs and indirect sources from degradation of semi-volatile precursor compounds (Buck et al., 2011). PFOA and PFNA are intentionally produced and because of this a large portion of the PFOA and PFNA found in the environment probably originates from direct sources (mainly the production process of fluoropolymers, (Prevedouros et al., 2006)) and subsequent waterborne transport to remote locations. Hence, sewage treatment plant effluent from industry or larger cities could represent hot-spots. In contrast, longer-chain perfluoroalkyl carboxylic acids (PFCAs) such as PFUnDA and PFTrDA are unintentionally produced substances, and their presence in the environment is probably due to both direct sources (impurities in PFOA and PFNA productions) and indirect sources such as atmospheric transport and degradation of precursors. The role of PFOS derivatives for the distribution and accumulation of PFOS in the environment is currently under investigation (Martin et al., 2010). FOSA is an intermediate product in the degradation of many PFOS precursors to PFOS, and is often analysed together with PFOS in environmental samples.
28.1.2 Toxicological Effects

Exponentially increasing concentrations of PFOS in wildlife were reported during the 1990s (Holmström et al., 2005). In biota, PFASs tend to accumulate in protein rich tissues such as blood, liver and eggs (Nordén et al., 2013). Toxic effects in laboratory experiments with mostly rodents include weight loss, liver enlargement, immunotoxicity and a number of developmental effects such as postnatal mortality (Lau et al. 2007). The common carp (Cyprinus carpio) exposed in lab to different PFOS concentrations experienced decreases in glycogen, and declines in condition factor and hepatosomatic index with increases in PFOS concentrations (Hagenaars et al., 2008). Due to large inter-species variations and even gender differences in toxicological effects, it is difficult to extrapolate observed effects to potential effects in humans. However, epidemiological studies on humans have increased in recent years. For instance, concentrations of PFOA in maternal blood and PFOA and PFOS in cord blood during pregnancy have been found to be negatively associated with birth weight (Apelberg B.J. et al., 2007, Fei et al., 2007), ponderal index, head circumference (Apelberg B.J. et al., 2007), and birth length (Fei et al., 2008b). In contrast, no associations between concentrations of PFOA and PFOS in maternal plasma during pregnancy and developmental milestones in early childhood have been found (Fei et al., 2008a). Current human exposure to PFASs is believed to be primarily the result of dietary intake (Vestergren and Cousins, 2009). Intake of contaminated fish from the Baltic Sea is one source of human exposure to PFASs (Berger U. et al., 2009).

28.1.3 Conventions, aims and restrictions

Perfluorooctane sulfonic acid, its salts, and perfluorooctane sulfonyl fluoride are among the nine new Persistent Organic Pollutants (POPs) included in The Stockholm Convention on POPs, an international agreement requiring measures for reducing or preventing release of dangerous substances to the environment. Production and use of PFOS is regulated in some countries (e.g., Canada and the EU), but large scale production continues in other parts of the world.

Due to their concentrations and/or temporal trends, PFOS, PFOA and PFNA are currently the PFASs of most concern for the Baltic Sea environment (HELCOM, 2010). Based on their documented relevance for the marine environment long-chain PFCAs and FOSA are additionally included in this report.

28.1.4 Target Levels

The target level used for PFOS in perch muscle is 9.1 μg/kg wet weight. For this report, we have used a conversion factor between liver and muscle, because PFOS is analysed in liver. This conversion factor is based on a study by Faxneld et al. (2014), and the new derived target level for PFOS in liver is 155 μg/kg wet weight. For further information on target levels and selection of target level see chapter 10.

28.2 Results

28.2.1 Spatial variation
The concentrations of PFHxA, PFHpA, PFOA, PFBS, PFHxS and PFDS (with the exception of few samples) were all under LOQ in perch liver during the period 2013–2015, and are therefore not reported in the maps below.

**Figure 28.1.** Spatial variation in concentration (ng/g wet weight) of PFOS in perch liver.
Figure 28.2. Spatial variation in concentration (ng/g wet weight) of FOSA in perch liver.

Figure 28.3. Spatial variation in concentration (ng/g wet weight) of PFNA in perch liver.
Figure 28.4 Spatial variation in concentration (ng/g wet weight) of PFDA in perch liver.

Figure 28.5 Spatial variation in concentration (ng/g wet weight) of PFUnDA in perch liver.
Figure 28.6 Spatial variation in concentration (ng/g wet weight) of PFDoDA in perch liver.

PFDoDA, perch liver

Figure 28.7 Spatial variation in concentration (ng/g wet weight) of PFTrDA in perch liver.

PFTrDA, perch liver
Figure 28.8 Spatial variation in concentration (ng/g wet weight) of PFTeDA in perch liver.

Figure 28.9 Spatial variation in concentration (ng/g wet weight) of PFPeDA in perch liver.
Of the fifteen PFASs that are measured, PFOS is the substance found in the highest concentrations in perch liver.

For PFOS (as well as for most other PFASs), a pattern of increasing concentration from north to the south western parts of Sweden is seen (Fig. 28.1), with exceptions for Lake Fysingen, Lake Krageholmsgjön, and Lake Limmingsjön with comparably high PFOS concentrations, probably originating from local sources. The highest concentration of PFOS in perch liver was found in Lake Fysingen close to Stockholm (83 ng/g ww) in 2013–2015, and the lowest concentration in Lake Spjutsjön (3.2 ng/g ww) in the County of Dalarna. Lake Fysingen is located close to the fire-fighting training sites at Rosersberg and also close to Arlanda airport, which most possibly explains the high concentrations.

The highest concentration of FOSA in perch liver in 2013–2015, 0.35 ng/g ww, was found in Lake Bysjön in the County of Värmland and the lowest concentration, <0.01 ng/g ww, were found in Lakes Allgjuttern in the County of Kalmar, Stor-Backsjön in the County of Jämtland, Fiolen in the County of Kronoberg, and Skärgölen in the County of Östergötland (Fig. 21.2).

The highest concentration of PFNA in perch liver in 2013–2015 was found in Lake Degervattnet in the County of Jämtland (2.4 ng/g ww) and the lowest concentration was found in Lake Svartsjön in the County of Västra Götaland (0.12 ng/g ww) (Fig. 28.3).

The highest concentration of PFDA in perch liver in 2013–2015, 4.0 ng/g ww, was found in Lake Fysingen in the County of Stockholm and Lake Bästeträsk in the County of Gotland. The lowest concentrations, 0.87 ng/g ww, were found in Lake Svartsjön in the County of Skåne (Fig. 28.4).

The highest concentration of PFUnDA in perch liver in 2013–2015 was found in Lake Hjärtsjön in Kronobergs County (11 ng/g ww) and the lowest concentration was found in Lake Krankesjön in the County of Skåne (1.3 ng/g ww) (Fig. 28.5).

The highest concentration of PFDoDA in perch liver in 2013–2015 (7.2 ng/g ww) was found in Lake Hjärtsjön in Kronobergs County and the lowest concentration (0.42 ng/g ww) was found in Lake Krankesjön in the County of Skåne (Fig. 28.6).

The highest concentration of PFTrDA in perch liver in 2013–2015, 17 ng/g ww, was found in Lake Hjärtsjön in Kronobergs County and the lowest concentration, 0.53 ng/g ww, was found in Lake Krankesjön in the County of Skåne (Fig. 28.7).

The highest concentration of PFTeDA in perch liver in 2013–2015 was found in Lake Hjärtsjön (4.1 ng/g ww) in Kronobergs County and the lowest concentration was found in Lake Krankesjön (0.15 ng/g ww) in the County of Skåne (Fig. 28.8).

The highest concentration of PFPeDA in perch liver in 2013–2015, 1.6 ng/g ww, was found in Lake Hjärtsjön in Kronobergs County and the lowest concentration, 0.096 and 0.10 ng/g ww respectively, were found in Lake Krankesjön in the County of skåne and in Lake Fysingen in the County of Stockholm (Fig. 28.9).

For most of the long-chain PFCAs, Lake Hjärtsjön in the County of Kronoberg had the highest concentrations compared to the other lakes.
28.2.2 Temporal variation
PFASs monitored within the programme but not presented as time trends are at concentrations below LOQ in the majority of the investigated years.

PFNA, PFDA, PFUnDA, and PFTraDA all showed significantly increasing concentrations of about 2.5–4.9 % per year in Arctic char liver from Lake Abiskojaure during 1980–2015 (Fig. 28.10–28.11). Furthermore, since 2006 downward trends were indicated for PFOS and PFDoDA in arctic char liver from Lake Abiskojaure (Fig. 28.10–28.11). For FOSA, a significant downward trend was seen since 2007 with an annual decrease of 22%. (Fig. 28.10). In perch liver from Lake Skärgölen increasing concentrations during 1980–2015 were seen for PFDA, PFUnDA, PFDoDA, and PFTraDA with about 3.0–6.0 % per year. In addition, since 2006 significant downward trends were indicated for PFNA, PFDoDA, and PFTraDA. A downward trend for PFOS in perch liver from Lake Skärgölen was seen for the whole time period with an annual mean of 2.9%, FOSA and PFUnDA in perch liver from Lake Skärgölen showed downward trends during the last nine years (Fig. 28.12–28.13).

The number of years required to detect an annual change of 10% for PFASs varies between 13–23 years for the char and perch time series.

**Figure 28.10.** PFOS, FOSA, PFNA, and PFDA concentrations (ng/g wet weight) in Arctic char liver from Lake Abiskojaure.
PFUnDA, PFDoDA and PFTrDA concentrations (ng/g wet weight) in Arctic char liver from Lake Abiskojaure.

PFOSA, PFOS, PFNA, and PFDA concentrations (ng/g wet weight) in perch liver from Lake Skårgölen.

Figure 28.11. PFUnDA, PFDoDA and PFTrDA concentrations (ng/g wet weight) in Arctic char liver from Lake Abiskojaure.

Figure 28.12. PFOS, FOSA, PFNA, and PFDA concentrations (ng/g wet weight) in perch liver from Lake Skårgölen.
Figure 28.13. PFUnDA, PFDoDA and PFTrDA concentrations (ng/g wet weight) in perch liver from Lake Skärgölen (1980–2013).

28.2.3 Comparison to threshold

The new derived target level of 155 ng/g wet weight in liver, based on the EQSbiota of 9.1 µg/kg wet weight in muscle, was not exceeded for any of the species and lakes.

28.3 Summary

The highest concentration of PFOS in perch liver (2013–2015) was found in Lake Fysingen, close to Stockholm. It is probably influenced by local fire-fighting activities and Arlanda airport.

For several of the PFASs (PFUnDA, PFDoDA, PFTrDA, PFTeDA, and PFPeDA), Lake Hjärtsjön in the County of Kronoberg had the highest concentrations in 2013.

PFNA, PFDA, PFUnDA, and PFTrDA all show significantly increasing concentrations in Arctic char liver from Lake Abiskojäure from 1980–2015. But, there are indications of downward trends for PFOS and PFDoDA in Arctic char since 2007, and the concentration of FOSA is decreasing during the same time period. In perch liver from Lake Skärgölen PFDA, PFUnDA, PFDoDA, and PFTrDA also show upward trends between 1980 and 2015. FOSA
and PFUnDA decrease during the most recent nine years and there are indications of decreasing trends for PFNA and PFDODA during the same time period. In addition, a decrease in PFOS concentration in perch liver from Lake Skärgölen is seen.

PFOS concentrations were below the suggested target level in all of the perch lakes.
29 PCDD/PCDF, Polychlorinated Dioxins and Dibenzo furans

Updated 16.10.31

29.1 Introduction

29.1.1 Uses, Production and Sources
Dioxins are unintentionally created during combustion of organic materials. They are highly toxic and carcinogenic.

Polychlorinated dibenzo-\(p\)-dioxins (PCDD) and dibenzofurans (PCDF) consist of 17 congeners considered to be of toxicological importance (210 congeners in total). PCDD/Fs are formed in several industrial processes and from most combustion processes (e.g., municipal waste incineration and small scale burning in poorly controlled conditions). The use of chlorine gas during pulp bleaching processes was formerly an important producer of PCDD/Fs.

Atmospheric deposition is the most important active source of PCDD/Fs to the Baltic Sea today (Sellström et al., 2009). The origin of the substances in air is, however, not fully known, although there are indications of strong impact of long-range atmospheric transport from south-western and southern Europe. It is also uncertain how historical emissions and secondary sources contribute, and how PCDD/Fs are accumulated in the food chain.

29.1.2 Toxicological Effects
The most relevant toxic effects of PCDD/Fs and dl-PCBs are developmental toxicity, carcinogenicity and immunotoxicity. The most relevant health effect seems to be developmental problems in children, where exposure can take place both during pregnancy and breast-feeding. A positive effect of restrictions and prohibitions of PCDD is that levels are decreasing in mother’s breast milk in Sweden (Norén and Meironyte, 2000, Fång et al., 2013).

29.1.3 Conventions, aims and restrictions
Releases of dioxins from industrial installations are mainly regulated by the IPPC Directive and the Waste Incineration Directive.

Dioxins are comprised by the objective of HELCOM’s strategy for hazardous substances that is to continuously reduce discharges, emissions and losses of hazardous substances, with a goal of their eventual cessation by the year 2020. The ultimate aim is to achieve concentrations in the environment near background values for naturally occurring substances and close to zero for man-made synthetic substances. This objective was adopted in 1998, and dioxins have been selected as one of the priority substances for immediate action.
PCDD/PCDF are part of the initial 12 Persistent Organic Pollutants (POPs) included in The Stockholm Convention on POPs, an international agreement requiring measures for reducing or preventing release of dangerous substances to the environment.

A dioxin and PCB strategy, including actions in the area of feed and food contamination and actions related to the environment including release reduction, was adopted by the EU in 2001.

29.1.4 Target Levels
The target level used for the sum of PCDD/F is 3.5 ng TEQ (2,3,7,8-tetrachlorodibenzo-p-dioxin equivalents) /kg wet weight, based on the WHO05-TEF (toxic equivalent factors) values (Van den Berg et al., 2006). However, due to back-compatibility in the time series the same target value of 3.5 ng/kg TEQ was used but calculated using the WHO98-TEF values (Van den Berg et al., 1998). The TEFs from 2005 do not differ to a high extent for the PCDDs and PCDFs compared to the TEFs from 1998 so, the WHO98-TEQs in the time series are therefore compared directly with the target level in WHO05 TEQ since older data on WHO05 TEQs is not available. The original target level has been recalculated for each time series based on the lipid percentage. The recalculated target level (Tv) together with the lipid percentage (lp) is shown above the statistical information in each time series. In addition, a second target value based on the EC quality standard for biota derived in regards to secondary poisoning (QSbiota, sec pois), set at 1.2 ng/kg TEQ wet weight is also used in this report. However, since the QSbiota, sec pois is based on the sum of PCDD/F and DL-PCBs and the times series only include PCDD/F concentrations, the target level was adjusted to correspond to PCDD/F part of Total TEQ target level (1.2 ng/kg wet weight). This was done by determining the ratio in TEQ concentrations between PCDD/F and Total TEQ (PCDD/F and DL-PCBs) of all herring samples during the last three years of monitoring (2011–2013). It was found that the contribution of PCDD/Fs to the Total TEQ of the samples was 53% on average. Hence the Target level for QSbiota, sec pois (excluding DL-PCBs) was set to 50% of the original value, i.e. the used value is 0.6 ng/kg TEQ wet weight. See chapter 10 for more information.

29.2 Results

29.2.1 Spatial variation
Figure 29.1 Spatial variation in concentration (pg/g lipid weight) of WHO05-TEQ (PCDD/PCDF) in perch muscle.

Figure 29.2. Spatial variation in concentration (pg/g wet weight) of WHO05-TEQ (PCDD/PCDF) in perch muscle.
For PCDD/PCDF, a pattern of increasing concentration from north to the southern parts of Sweden is seen (Fig. 29.1–29.2). The spatial pattern for perch is consistent with the spatial pattern for pike (Fig. 29.4, 29.6). The levels of TCDD-equivalents in pike from Bolmen is more than three times higher than in pike from Storvindeln, which is located further north.

The highest concentrations on a lipid weight basis of TCDD-equivalents (WHO05) in 2013-2015, 30 pg/g lw and 26 pg/g lw respectively, in perch muscle were found in Lake Sännen in the County of Blekinge and Lake Stora Skärssjön in the County of Halland (Fig. 29.1) in 2013–2015. The lowest concentration (3.6 pg/g lw) in perch during the same period was found in Lake Remmarsjön in the County of Västernorrland (Fig. 29.1).

29.2.2 Temporal variation

Temporal trends of TCDD and TCDF on a lipid weight basis in pike from Lake Bolmen and in perch from Lake Skärgölen show similar patterns (Fig. 29.4, 29.8). TCDD and TCDF have decreased significantly during the period 1967/1980–2015, with an annual change of about 1.5-2.7% for TCDD and about 1.4-1.8% for TCDF. In pike from Lake Storvindeln TCDD is decreasing with 5.4% per year while no trend is seen for TCDF on a lipid weight basis (Fig. 29.6).

Temporal trends of TCDD-equivalents for dioxinlike PCB on a lipid weight basis are decreasing significantly in pike from Lake Storvindeln and Lake Bolmen, and in perch from Lake Skärgölen during the period 1980/1990–2015 of about 2.3–6% (Fig. 29.4, 29.6 and 29.8).

The number of years required to detect an annual change of 10% for PCDD/PCDF varies between 12–14 years for the pike and perch time series (Fig. 29.3–29.8).

Figure 29.3 PCDD/PCDF concentrations (pg/g wet weight) in pike muscle from Lake Bolmen. The TCDD-EQVs are calculated using the WHO98 TEF. The green area denotes the levels below the target level for PCDD/Fs in fish (set for human consumption) and the blue area denotes the levels below QSbiota (secondary poisoning). The TCDD-EQV is calculated using the WHO98-TEF.
Figure 29.4. PCDD/PCDF concentrations (pg/g lipid weight) in pike muscle from Lake Bolmen. The TCDD-EQVs are calculated using the WHO98 TEF. The green area denotes the levels below the target level for PCDD/Fs in fish (set for human consumption) and the blue area denotes the levels below QSbiota (secondary poisoning). The TCDD-EQV is calculated using the WHO98-TEF.

Figure 29.5. PCDD/PCDF concentrations (pg/g wet weight) in pike muscle from Lake Storvindeln. The TCDD-EQVs are calculated using the WHO98 TEF. The green area denotes the levels below the target level for PCDD/Fs in fish (set for human consumption) and the blue area denotes the levels below QSbiota (secondary poisoning). The TCDD-EQV is calculated using the WHO98-TEF.
Figure 29.6. PCDD/PCDF concentrations (pg/g lipid weight) in pike muscle from Lake Storvindeln. The TCDD-EQVs are calculated using the WHO98 TEF. The green area denotes the levels below the target level for PCDD/Fs in fish (set for human consumption) and the blue area denotes the levels below QSbiota (secondary poisoning). The TCDD-EQV is calculated using the WHO98-TEF.

Figure 29.7. PCDD/PCDF concentrations (pg/g wet weight) in perch muscle from Lake Skärgölen. The TCDD-EQVs are calculated using the WHO98 TEF. The green area denotes the levels below the target level for PCDD/Fs in fish (set for human consumption) and the blue area denotes the levels below QSbiota (secondary poisoning). The TCDD-EQV is calculated using the WHO98-TEF.
29.2.3 Comparison to thresholds

In all areas and species, TCDD-equivalent concentration is below the suggested target level based on the EC EQS (Environmental Quality Standard) of 3.5 pg WHO05-TEQ/g wet weight and the QSbiota, sec pois (excluding DL-PCBs) of 0.6 pg WHO05-TEQ/g wet weight.

29.3 Summary

The highest concentrations of TCDD-equivalents in perch muscle were found in Lake Sännin in the County of Blekinge and Lake Stora Skärtsjön in the County of Halland in 2012-2015.

PCDD/PCDF concentrations varied between species and sites; however temporally, the concentration of TCDD has decreased at the three sampling sites, while TCDF has only decreased at the two southern sites (i.e. Bolmen and Skärgölen) and the concentration of TCDD-equivalents for dioxinlike PCB has decreased in perch from Lake Skärgölen, and in pike from Lake Storvindeln and Lake Bolmen. Also the concentration of TCDD-equivalents for PCDD/PCDF in pike from Lake Storvindeln has decreased.

In all areas and species, TCDD-equivalent concentration (PCDD/PCDF) is below the suggested target level.
30 Polybrominated flame retardants

30.1 Introduction

30.1.1 Uses, Production and Sources
Polybrominated diphenyl ethers (PBDEs) are produced as three different technical products; penta-, octa- and decaabromo diphenyl ether (BDE). Each of these products includes a few major congeners. For pentaBDE these are BDE-47, -99, and -100. OctaBDE contains mainly BDE-183, while decaBDE includes almost exclusively BDE-209 (La Guardia et al., 2006). Hexabromocyclododecan (HBCDD) is produced as a mixture of three stereoisomers – α-, β- and γ-HBCDD (Covaci A. et al., 2006). Both PBDEs and HBCDD are used as additive flame retardants incorporated into materials such as plastics and textiles.

PBDEs leak into the environment during production, use, or disposal of such products. PBDEs are mainly spread via diffuse distribution in the atmosphere and in rivers. HBCDD is bioaccumulative, lipophilic and persistent, and accumulates in the food-web.

More comprehensive information concerning PBDEs and HBCDD, e.g. sources and environmental fate, can be found in the extensive reports from the European Food Safety Authority (EFSA), on PBDEs (EFSA, 2012) and HBCDD (EFSA, 2011).

30.1.2 Toxicological effects
Several PBDE congeners and HBCDD have been shown to cause neurotoxic effects in rats and mice. In mammals, behaviour, learning (Eriksson et al., 2006a, Eriksson et al., 2006b) and affects on hormonal functions have been reported (Legler, 2008). Animals exposed to PBDEs and HBCDD during a sensitive stage of brain development have later shown reduced memory and learning disabilities (Viberg, 2004, Eriksson et al., 2006a, Eriksson et al., 2006b). In birds, several aspects regarding reduced reproductive success has been documented (Fernie et al., 2009). Chemical products and goods containing concentrations over a certain level of these BDEs are banned in the EU. Brominated flame retardants (BFR) are also considered to be endocrine disruptors, and in particular, effects on the thyroid hormone system are seen (Darnerud, 2008, UNEP and WHO, 2013).

30.1.3 Conventions, aims and restrictions
The PBDEs tetrabromodiphenyl ether, pentabromodiphenyl ether, hexabromodiphenyl ether and heptabromodiphenyl ether are among the nine new Persistent Organic Pollutants (POPs) included in The Stockholm Convention on POPs. Within the EU, the penta- and octaBDE products were banned for use in 2004.

A Swedish ban of decaBDE was established in 2007, but this ban was withdrawn when decaBDE was included in the RoHS directive in 2008. PBDEs are also on the list of prioritized substances within the Water Framework Directive (The European Commision, 2008b).
Since the autumn of 2014 HBCDD is included in The Stockholm Convention on POPs (Stockholm Convention).

**30.1.4 Target Levels**

The target level (TL) used for HBCDD in the time series for fish is 167 µg/kg wet weight. The target level used for the sum of BDE-28, -47, -99, -100, -153 and -154 for fish is 0.0085 µg/kg. For further information on TL and selection of target level see chapter 10. The recalculated target level (Tv) together with the lipid percentage (lp) is shown above the statistical information in each time series.

**30.2 Results**

The concentration of HBCDD is under LOQ in a majority of the samples for all matrices and sites and is therefore not presented in the spatial maps and time trends.

**30.2.1 Spatial variation**

Figure 30.1. Spatial variation in concentration (ng/g lipid weight) of BDE-47 in perch muscle.
Figure 30.2. Spatial variation in concentration (ng/g lipid weight) of BDE-99 in perch muscle.

Figure 30.3. Spatial variation in concentration (ng/g lipid weight) of BDE-100 in perch muscle.
Figure 30.4. Spatial variation in concentration (ng/g lipid weight) of BDE-153 in perch muscle.

Figure 30.5. Spatial variation in concentration (ng/g lipid weight) of BDE-154 in perch muscle.
BDE-47, -99, -100, -153 and -154 generally show higher concentrations in perch muscle from lakes in the Stockholm area and in the southern part of Sweden compared to the northern part, with a few exceptions.

Of the five PBDEs that are measured, BDE-47 is the substance found in the highest concentrations in perch muscle.

The highest concentration of BDE-47 in perch muscle was found in Lake Stora Skärsjön in the County of Halland (34 ng/g lw) in 2013–2015, and the lowest concentration in Lake Svartsjön (0.74 ng/g lw) in the County of Södermanland (Fig. 30.1). The highest concentration of HBCDD in perch muscle was found in Hjärtsjön in the County of Kronoberg (5.9 ng/g lw) and the lowest concentration was found in Skärgölen in the County of Kalmar (1.4 ng/g lw).

### 30.2.2 Temporal variation

No general linear trend is observed during the whole monitoring period for the PBDEs consistent in both Arctic char from Lake Abiskojaure (1980–2015), and in pike from Lake Bolmen (1966–2015). The concentrations of PBDEs in Lake Bolmen increased from the start of the monitoring period until the late 80s to the mid 90s with an annual increase of about 10–19% and appear to have decreased since then. The lower brominated PBDEs, i.e. tetra- and penta-BDEs (BDE-47, -99 and -100), peaked earlier than the higher brominated PBDEs, i.e. hexa-BDEs (BDE-153 and -154) (Fig. 30.7–30.10). The concentrations are still at higher levels compared to the start of the monitored period.
The number of years required to detect an annual change of 10% for PBDEs varies between 14–29 years for the char and pike time series.

Figure 30.7. BDE-47, -99 and -100 concentrations (ng/g lipid weight) in Arctic char muscle from Lake Abiskojaure.

Figure 30.8. BDE-153 and -154 concentrations (ng/g lipid weight) in Arctic char muscle from Lake Abiskojaure.
Figure 30.9. BDE-47, -99 and -100 concentrations (ng/g lipid weight) in pike muscle from Lake Bolmen.

Figure 30.10. BDE-153 and -154 concentrations (ng/g lipid weight) in pike muscle from Lake Bolmen.
30.2.3 Comparison to thresholds
Since data on BDE-28 is lacking for most years, it is not possible to compare the sum of PBDEs against the target level. However, in all areas, the concentration of BDE-47 congener is by itself above the suggested target level for the sum of BDE-28, -47, -99, -100, -153 and -154 of 0.0085 µg/kg wet weight.

30.3 Summary
BDE-47, -99, -100, -153 and -154 generally show higher concentrations in perch muscle from lakes in the Stockholm area and in the southern part of Sweden, with a few exceptions.

The highest concentration of HBCDD in perch muscle was found in Hjärtsjön in the County of Kronoberg, however, the concentration of HBCDD is under LOQ in a majority of the samples for all species and sites.

The highest concentration of BDE-47 in perch muscle was found in Lake Stora Skärsjön in the County of Halland in 2013–2015.

No general linear trend is observed during the whole monitoring period for the PBDEs in Arctic char from Lake Abiskojaure and in pike from Lake Bolmen. Moreover, the concentrations of PBDEs in Lake Bolmen increased from the start of the monitoring period until the late 80s to the mid 90s and appear to have decreased since then. The lower brominated flame retardants (BDE-47, -99 and -100) peaked earlier than the higher (BDE-153 and -154).

In all areas, the BDE-47 concentration is above the suggested target level for the sum of BDE-28, 47, -99, -100, -153 and -154 of 0.0085 ng/g wet weight.
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