report

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Risk assessment of sedimentary mercury at Snäckviken, Södertälje channel

John Sternbeck John Munthe Tomas Viktor

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IVL Svenska Miljöinstitutet AB

IVL Swedish Environmental Research Institute Ltd.

Adress/address

Box 21060
100 31 Stockholm

Telefonnr/Telephone

08-598 563 00

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Report Summary

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John Sternbeck, John Munthe and Tomas Viktor

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Risk assessment of sedimentary mercury at Snäckviken, Södertälje channel

Sammanfattning/Summary

The sediments in the northern Södertälje channel are highly polluted with mercury (Hg). Although point source emissions stopped during the mid 1960ies, considerable amounts of Hg are still present in the upper sediments. Because Hg is inherently toxic and may biomagnify, these sediments compose a potential hazard to the local ecosystem. There is also a risk that Hg is transported downstream into the Baltic coast. This report provides a risk assessment based on all pertinent data from the region. The report is intended to provide a basis for discussing whether remediation should be considered or not. The ultimate goal is to identify whether Hg is causing adverse biological effects and, if this is the case, to suggest how exposure of Hg to biota can be limited. In such cases the exposure pathway must be identified. The major overall conclusions of this risk assessment are: Sedimentary mercury in Snäckviken causes no significant ecological impact *in situ*.

Dredging in the area leads to elevated risk in the short term.

Intensified vessel traffic may bring about a permanently elevated risk.

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Sammanfattning

Sedimenten i norra Södertälje kanal är starkt förorenade med kvicksilver (Hg). Trots att punktkällans utsläpp upphörde i mitten av 1960-talet så innehåller de övre sediment-skikten avsevärda mängder Hg. Kvicksilver är toxiskt och kan biomagnifieras varför dessa sediment utgör en potentiell risk för det lokala ekosystemet. Dessutom finns en potentiell risk att Hg transporteras ut i Östersjön.

Södertälje kanal är viktig farled för större fartyg (upp till 130 m) som går in till Mälaren, vilket påverkar den risk som Hg i sedimenten utgör på två sätt: (1) Fartygen orsakar resuspension varigenom Hg sprids geografiskt, och (2) Kanalen måste muddras vid regelbundna intervall för att bibehålla farledsdjupet. Muddringen leder till lokal spridning av Hg men kan även ge betydande ekologisk påverkan om muddermassorna dumpas i en djuphåla.

I föreliggande rapport redovisas en riskbedömning baserad på alla relevanta data från området. Rapporten syftar till att utgöra ett underlag för diskussioner om huruvida åtgärder bör övervägas eller ej. Det övergripande syftet är att avgöra om Hg orsakar ekologiska effekter och, om detta är fallet, föreslå hur biotans exponering för Hg kan minskas. I dessa fall måste exponeringvägarna identifieras.

Att identifiera ekologiska effekter orsakade av kvicksilver är inte trivialt eftersom (1) effekter av Hg tenderar att uppstå högst upp i näringskedjan och kan därför vara svåra att hänföra till ett visst begränsat område; (2) ett flertal miljöföroreningar kan orsaka liknande effekter eller samverka; (3) variationer i artsammansättningen, t.ex. minskad diversitet, kan bero på ett antal miljöfaktorer förutom miljögifter. Biologiska effekter kan inte heller förutspås enbart genom att jämföra Hg-halter i sediment med generella riktvärden; biotillgänglighet och exponeringsvägar måste också utvärderas. Biotillgängligheten av sedimentbundet Hg varierar markant med sedimentens egenskaper, varför generella riktvärden har litet värde vid bedömning av risken för eventuella effekter. Huvudslutsatserna i denna riskbedömning är:

- Kvicksilver i Snäckvikens sediment orsakar ingen betydande ekologisk påverkan *in situ*.
- Muddring ökar risken på ett par års sikt.
- Utökad fartygstrafik skulle kunna medföra en permanent förhöjd risk, beroende på de hydrologiska effekterna orsakade av fartygen.
- Ur vetenskaplig synvinkel rekommenderar vi att sedimenten ligger kvar, men att varje ingrepp i området föregås av noggranna miljöbedömningar och följs genom kontrollprogram.

De övergripande resultaten är:

- Sedimenten är starkt förorenade med Hg, och Snäckviken har den högsta föroreningsgraden. Kemiska lakningar tyder på att biotillgängligheten är låg.
- Koncentrationerna av total-Hg i vatten är ca 2-3 gånger högre vid Linasundet och Snäckviken jämfört med regionala bakgrundsvärden. Metyl-Hg i vatten är nära eller lägre än detektionsgränsen (0.06 ng/l), dvs. mindre än 3% av total-Hg.
- Muddring i farleden medförde en 30-faldig ökning av total-Hg i Snäckvikens bottenvatten. Även sedimentationsflödet av Hg ökande under muddring.
- En exponeringsanalys indikerar att resuspension är den dominerande spridningsmekanismen under icke-muddringsperioder. Resuspension orsakas sannolikt till stor del av fartygstrafiken.
- Halterna av metyl-Hg i zooplankton och zoobentos är låga respektive måttligt förhöjda, trots mycket höga halter i sedimenten.
- Sammanfattningsvis så tyder de låga halterna av Hg i biota och vatten på låg biotillgänglighet och låg mobilitet av det sedimentbundna Hg.
- Den bentiska faunan uppvisar mycket låga index i Snäckviken, men indexen är inte korrelerade med Hg i sedimenten. Eventuella effekter av Hg döljs möjligen av andra miljögifter såsom PAH, PCB och petroleumkolväten, men störningarna kan även bero på fartygstrafiken.
- Unga abborrar (1+) uppvisar något högre Hg-halter i Linasundet och Snäckviken jämfört med Kiholmen under normala förhållanden. Vid muddring ökade Hg i unga abborrar fyrfaldigt.
- Abborre reproducerar sig normalt i Snäckviken och överlevnad och status av embryo och larv är jämförbart med referensområden.

Summary

The sediments in the northern Södertälje channel are highly polluted with mercury (Hg). Although point source emissions stopped during the mid 1960ies, considerable amounts of Hg are still present in the upper sediments. Because Hg is inherently toxic and may biomagnify, these sediments compose a potential hazard to the local ecosystem. There is also a risk that Hg is transported downstream into the Baltic coast.

Södertälje channel is also an important route for large ships (up to 130 m) going into lake Mälaren. This fact has two important consequences for the potential risk of sedimentary Hg: (1) The ships cause sediment resuspension whereby Hg is spread over a larger area and is brought in contact with biota, and (2) The channel must dredged at regular intervals. Dredging causes dispersion of Hg at the dredging site but may also cause significant ecological impact if the dredged sediments are dumped in a deep.

This report provides a risk assessment based on all pertinent data from the region. The report is intended to provide a basis for discussing whether remediation should be considered or not. The ultimate goal is to identify whether Hg is causing adverse biological effects and, if this is the case, to suggest how exposure of Hg to biota can be limited. In such cases the exposure pathway must be identified.

Field identification of biological effects caused by Hg is not trivial because (1) effects of Hg tend to occur in the top of the food-chain and may thus be difficult to link to a certain restricted area; (2) numerous pollutants may cause similar effects or act synergistically; (3) variations at the community level, e.g. low diversity, are caused by numerous environmental factors other than pollutants. Neither can biological effects be predicted solely based on Hg concentrations in sediments and general guidevalues; bioavailability and exposure pathways must also be considered. The bioavailability of sedimentary Hg varies significantly with sediment conditions, why general guidevalues for total-Hg are of little use for assessing the risk of adverse effects.

The major overall conclusions of this risk assessment are:

- Sedimentary mercury in Snäckviken causes no significant ecological impact *in situ*.
- Dredging in the area leads to elevated risk in the short term.
- Intensified vessel traffic may bring about a permanently elevated risk, but is dependent on the hydrological impact of the vessels.
- From a scientific point of view we recommend not to dredge the sediments, but that all operations in the area are preceded by careful environmental assessments and are followed through control programmes.

These are the major results:

- Sediments show extensive contamination of Hg, with Snäckviken as the generally most contaminated area. Chemical leachings indicate that the bioavailability is low.
- The concentration of total-Hg in water is ca 2-3 times higher at Linasundet and Snäckviken compared to the regional background level. The methyl-Hg fraction measured in the water column is close to or below detection limit (0.06 ng/l), i.e. less than 3% of total-Hg.
- During dredging operations in the navigation channel, a 30-fold increase of the total-Hg concentration was detected in bottom water in Snäckviken. Also the settling fluxes of Hg increased during dredging.
- Exposure analysis indicates that resuspension is the major dispersion mechanism during non-dredging conditions. The passing of large vessels is most likely the major cause of resuspension.
- The methyl-Hg content in zooplankton and zoobenthos is low and moderately elevated, respectively, despite the very high Hg concentrations in the sediments.
- Overall, concentrations of Hg in biota and the water column indicate low bioavailability and low mobility of sedimentary Hg.
- The zoobenthic fauna community showed very low indexes in Snäckviken, but no positive correlation to Hg content in sediments could be established. Possible effects of Hg may be obscured by the presence of other pollutants such as PAH and PCB, and petroleum compounds, but the low indexes may also be caused by physical effetcs from thevessel traffic.
- Small perch (1+year) showed slightly higher Hg content in Linasundet and Snäckviken compared to Kiholmen during normal conditions. During dredging operations, the Hg content in small perch increased 4-fold.
- Perch successfully reproduce in the Snäckviken area and the embryo/larvae survival and status is comparable with reference sites.

1 Background

1.1 Regional setting

Approximately 2.5 % of the annual outflow of Lake Mälaren goes through Södertälje channel (Figure 1). A lock in the city of Södertälje regulates the border between Lake Mälaren and the brackish Baltic Sea. The average water flow is ca 4×10^5 m^3/d (= 4.5 m^3/s) which goes through a culvert in parallel with the channel. Lesser amounts go directly through the lock (ca 2.5×10^4 m^3/d). The area receives no industrial or municipal effluents today.

The boat traffic through the area is intensive. An average of 12 large vessels a day are passing the area either north or southbound. The minimum depth in the channel (7.6 m) and the lock construction provides an upper limit on the size of the incoming vessels to 130 m and ca 14.000 tonne.

The aquatic life in the area is not so well documented compared to all sediment-characterisations that have been conducted in the area (chapter 3.2). In the investigations in the late 90's, concerning dredging in the channel system, occurrence and abundance of zooplanktonic, zoobenthic and fish fauna was included in the programme (Parkman 1998; Viktor 2000a; Sternbeck 2000).

The occurrence and abundance of zooplankton vary with the season. The reproduction period starts in early spring and proceeds during the summer and fall. Plankton sampled in spring, summer and fall 1996-97 show a similar distribution of groups in Snäckviken and outside Kiholmen. The major group found in the zooplankton samples were Copepods, independent of the season.

The zoobenthic fauna in Snäckviken is described in detail in Sternbeck et al. (2000). Benthic fauna was also sampled in the northern part of Linasundet and at Kiholmen during 1996 and 1998 (see Parkman 1998, Viktor 2000a) and an additional sampling including Snäckviken was performed in 2000 (Viktor, 2000b). The benthic fauna structure is described in chapter 6. Several benthic organisms which are indicators of a good water quality were found in the littoral zone of Krutholmen, e.g. insectslarvae representing family *Ephemeroptera, Limnophilus* and *Odonata*. Perch (Perca fluviatilis) and roach (Rutilus rutilus) dominated the fish catched in Linasundet, with occasional occurrence of burbot (*Lota lota*) and smelt (*Osmerus eperlanus*). In Snäckviken the fish catch was dominated by perch with occasional occurrence of roach and smelt. All these findings indicate good water quality.

There are a few species of birds in the area that may be exposed to mercury since they represent the top in the local foodchain. The entire common top consumers of fish, i.e Osprey (*Pandion haliaetus*), Heron (*Ardea cinerea*), great crested grebe (*Podiceps cristatus*) and goosander (*Mergus merganser*) are present and feed and breed in the area. The opportunistic common gull (*Larus canus*) breeds for example at the island of Krutholmen with app.10 pairs every year.

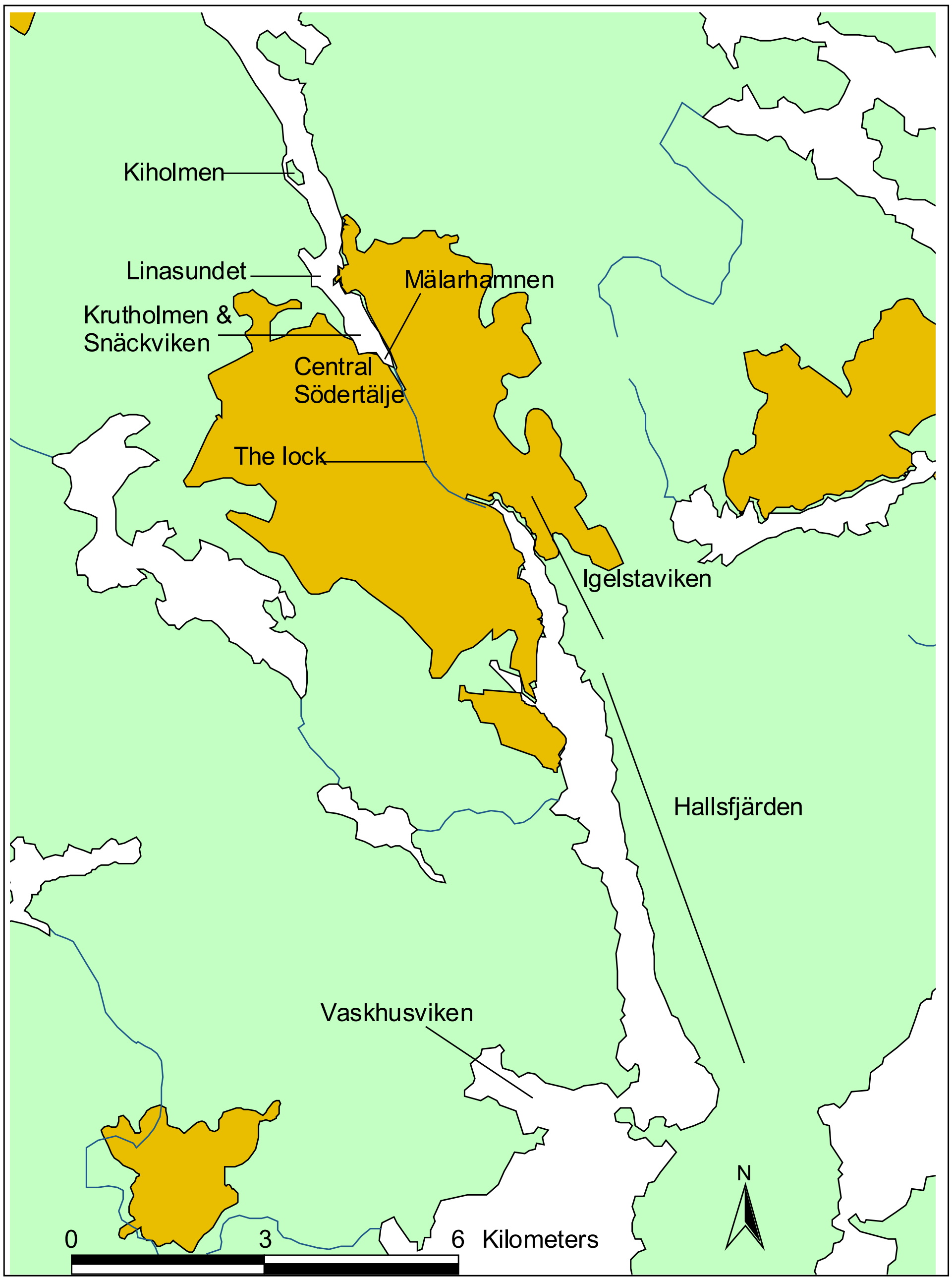

Figure 1. Overview of the area with geographical names used in the report shown. Lake Mälaren is north of the lock and the Baltic Sea is south of the lock.

1.2 Previous studies

A number of studies related to the presence of Hg in the Södertälje channel have been performed over the years. For reference, their contents are summarized in Table 1.

Table 1. Overview of available data from studies in the Mälarregion of Södertälje channel.

Report	Mercury concentrations			Other data	Area
	Sediment	Water	Biota		
Sternbeck et al. (2000)	Hg			zoobenthos	Lake Mälaren
Viktor (2000b)			Hg meHg		Lake Mälaren
Viktor (2000a)	Hg	Hg, meHg	Hg	other metals & organic pollutants in sediments; ecotoxicological characteri-sation of sediments	Lake Mälaren & Hallsfjärden
Sternbeck and Östlund (1999)	Hg, meHg, phenyl-Hg			bioavailability; chronology	Lake Mälaren & Hallsfjärden
Parkman et al. (1998)	Hg, meHg	Hg, meHg	Hg, meHg	other metals & organic pollutants in sediments; sediment traps; ecotoxico-logical characterisation of sediments	Lake Mälaren & Hallsfjärden
Skoglund (1998)	Hg meHg	Hg meHg	Hg	turbidity and suspended soil during and after dredging and dumping operations	Lake Mälaren & Hallsfjärden
ELK (1993) 2 st	Hg		Hg	other metals & organic pollutants in sediment	Lake Mälaren & Hallsfjärden
VBB (1972)	Hg				
IVL (1970)	Hg				Lake Mälaren

List of available reports.

Sternbeck J., Munthe J., Liljeberg M. and Brown R. (2000) Mercury and zoobenthos in the sediments in Snäckviken, Södertälje. IVL A20005, 2000-01-17. For AstraZeneca.

Sternbeck J. and Östlund P. (1999) Kvicksilver i sediment i Södertälje kanal: mängder, förekomstformer och biotillgänglighet. IVL A990290, 1999-11-03. For AstraZeneca (in Swedish)

Parkman H., Remberger M. and Viktor T. (1998) Miljöövervakningsprojekt i Södertälje kanalområde för bedömning av effekterna från muddring och tippning av förorenade sediment – lägesrapport 1997. IVL B1290. (in Swedish)

Viktor T. (2000a) Miljöövervakningsprojekt i Södertälje kanalområde … – delrapport 2 lägesrapport 1998: Bedömning av vattenkvalitet samt biologiska effekter i recipienten samt i akvatiska testsystem. Ett år efter avslutade muddrings- och tippningsarbeten. IVL B 1357 (in Swedish)

Viktor T. (2000b) Undersökning av kvicksilver halter i fisk och bottendjur insamlade i Mälaren norr om Södertälje. IVL Svenska Miljöinstitutet AB, 2000-11-29. For AstraZeneca. (in Swedish)

Skoglund P-O and Torstensson H (1998) Muddringsarbeten i Södertälje kanal och Mälaren 1996/97 KM Lab rapport till Sjöfartsverket (in Swedish)

ELK AB Rapporter till Sjöfartsverket a. 1993-07-07 b. 1993-12-03 (in Swedish)

IVL(1970) Kvicksilverhalter i bottensediment från Mellanfryken, Övre Hillen och Södertäljeviken. IVL B62. (in Swedish)

VBB (1972) see Andersson L. and Danielsson M (1997) Sammanställning av kvicksilverundersökningar i Södertälje kanal och Hallsfjärden (in Swedish)

1.3 Dredging

Södertälje channel is an important navigation channel for large ships going into Lake Mälaren (see chapter 3.1). To maintain the regulated depth in the navigation channel it is necessary to complement dredge approximately every decade. The complement dredgings are often necessary at the outer side of bends, i.e Linasundet and in Mälarhamnen because of the erosion and resedimentation processes.

1.3.1 Future scenario, plans and actions in the area

A brief summary from the discussion held in Södertälje with Rolf Bertilsson (director of Lake Mälarens maritime traffic area) and Tomas Åhsberg (Environmental department, Norrköping) from the Swedish maritime administration.

Scenario 1 No reconstruction of the lock

Additional dredgings were made in Mälarhamnen, Snäckviken and in Linasundet during the winter 1996-97. The dredgings in the navigation channel secured the regulated depth in the area up to Lake Mälaren. Annual depth measurements have stated that no erosion or sedimentation have occurred in the navigation channel, so far. The depth measurements will continue on a yearly basis. If nothing exceptional occur in the area, an additional dredging will be necessary during 2006-07. The frequency of additional dredgings is approximately 10 years within the channel area, if the vessel traffic stays in a status quo level, as today.

Scenario 2 Construction of a new lock

Local authorities in the whole Mälar region and governmental departments have discussed a widening of the lock in Södertälje. The reconstruction of the lock will allow vessels up to 180 m to passage trough the lock. The main reason is to provide larger ships up to 25.000 tonnes to enter Lake Mälaren. The eventual need of larger transport vessels carrying wood for heat production depends especially on the forthcoming energy policy for the Mälar region. A main goal is to minimize the use of fossil fuel and use renewable energy sources in local combustion plants. If the authorities decide to increase the use of firewood in the combustion plants, then there will be a regional shortage situation of wood. The pulp and paper and saw industries can not purchase enough raw material in the region already today, so a lot of the timber used in the industries had to be imported. The Baltic states and the Russian federation are probably the market that can offer the cheapest wood in sufficient quantities. The traffic through the Södertälje channel will then increase with big Baltic and Russian riverboats carrying combustion wood for power plants. The schedule for a decision of a reconstruction of the lock is several years ahead, according to the Swedish maritime administration.

2 Degree of contamination

This chapter reviews the degree of contamination of Hg in sediments, water column and biota. When present, regional background values are also shown. Other pollutants have not been not so comprehensively studied but their existence is briefly outlined. Industrial emissions of mercury occurred until the mid 1960-ies, when production of the Hg-containing fungicides ceased. Emissions are assumed to have taken place in Snäckviken, south of Krutholmen. Other sources of Hg in the region may possibly have contributed to the high background levels of sedimentary Hg in the region. Lake Turingen is severely polluted with Hg and is located upstream, and its outlet reaches Lake Mälaren ca 8 km northwest of Kiholmen.

2.1 Sediments

Sediments from more than 70 stations have been analysed at various depths for total-Hg (n=241) and occasionally for methyl-Hg. The investigated area begins north of Kiholmen and goes south to Igelstaviken, with most samples taken in Snäckviken. Total-Hg spans over a range of more than 5 orders of magnitude. Values of total-Hg are statistically represented in figure 2, showing that concentrations generally are higher in Snäckviken compared to all data, and that the concentrations have declined somewhat toward the sediment surface. Figure 2 also shows that several extremely high values are found outside Snäckviken.

The geographical distribution of total-Hg in surface sediments is shown in figure 3. All but one station in Snäckviken have Hg concentrations that are considered as very high according the Swedish Environmental Quality Criteria, i.e. > 5 µg/g dw (Swedish EPA, 1999a). Most stations outside Snäckviken have high concentrations (1-5 µg/g) according to the same criteria. However, these criteria are not based on ecological risks, but relate to the statistical distribution of Hg concentrations in Swedish lake sediments. Risk-based guidevalues are mentioned in chapter 6.1.

It is difficult to choose a representative regional background value, because the entire Mälar region is to some extent influenced by local/regional anthropogenic activities. The upstream stations north of Kiholmen, that are most distant to the probable point source, show Hg concentrations of 3-4 µg/g in the surface sediments, whereas the background value for small lakes in southern Sweden is 0.16 µg/g (Swedish EPA, 1999a). As another regional reference, surface sediments in central Stockholm show a median value of 1.9 µg/g (Östlund et al., 1998).

The distribution of methyl-Hg in surface sediments is shown in figure 4. Very few studies are available for comparison but the levels appear high (cf. Parkman, 1993; Verta and Matilainen, 1995). Methyl-Hg constitutes 0.05-1 % of total-Hg, which is normal for sediments. Concentrations generally decrease with sediment depth although the three highest values (>100 ng/g) occur at depths larger than 10 cm.

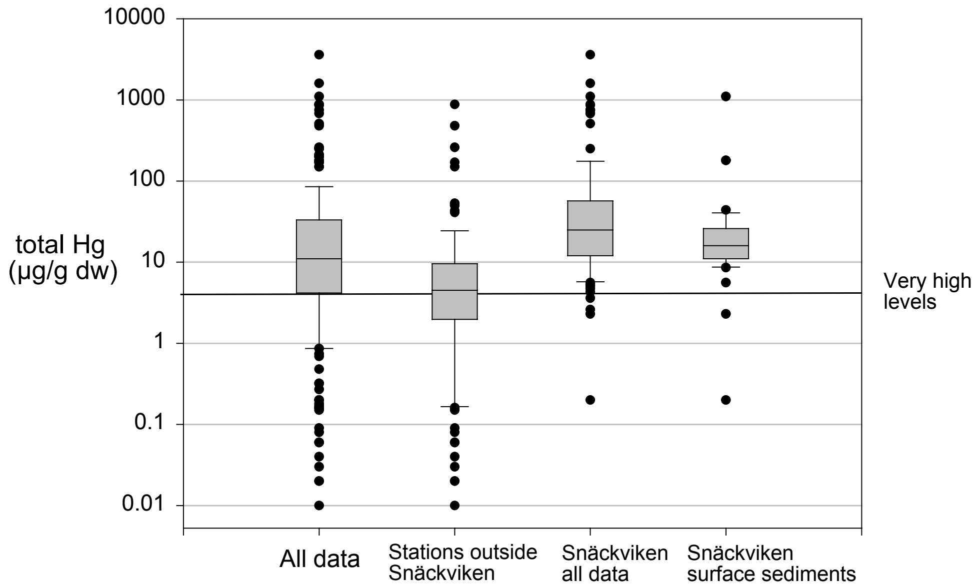

Figure 2. Distribution of sedimentary Hg concentrations. The boxes represent median values, 25- and 75-percentiles. The whiskers represent 10- and 90 percentiles and the dots represent all outliers. Values are not normally distributed. A line is inserted representing the value above which concentrations are classified as being very high (Swedish EPA, 1999a).

2.1.1 Speciation and bioavailability

In attempt to estimate the bioavailability of sedimentary Hg, i.e. how easily Hg detaches from sediment particles to solution, four different chemical leachings were performed on sediments from five stations (Sternbeck and Östlund, 1999). These studies also indicate if Hg is likely to be mobilised from sediments to bottomwater. Total-Hg was very strongly bound and neither acidic, weakly oxidising nor reducing conditions released significant amounts. Methyl-Hg was slightly more leachable (~bioavailable) than total-Hg, but showed the same general pattern. Both total-Hg and methyl-Hg appeared to a large extent in the humic fraction. The bioavailability of sedimentary Hg is also indicated by a few analyses of the Hg levels in benthic fauna (chapter 4.3).

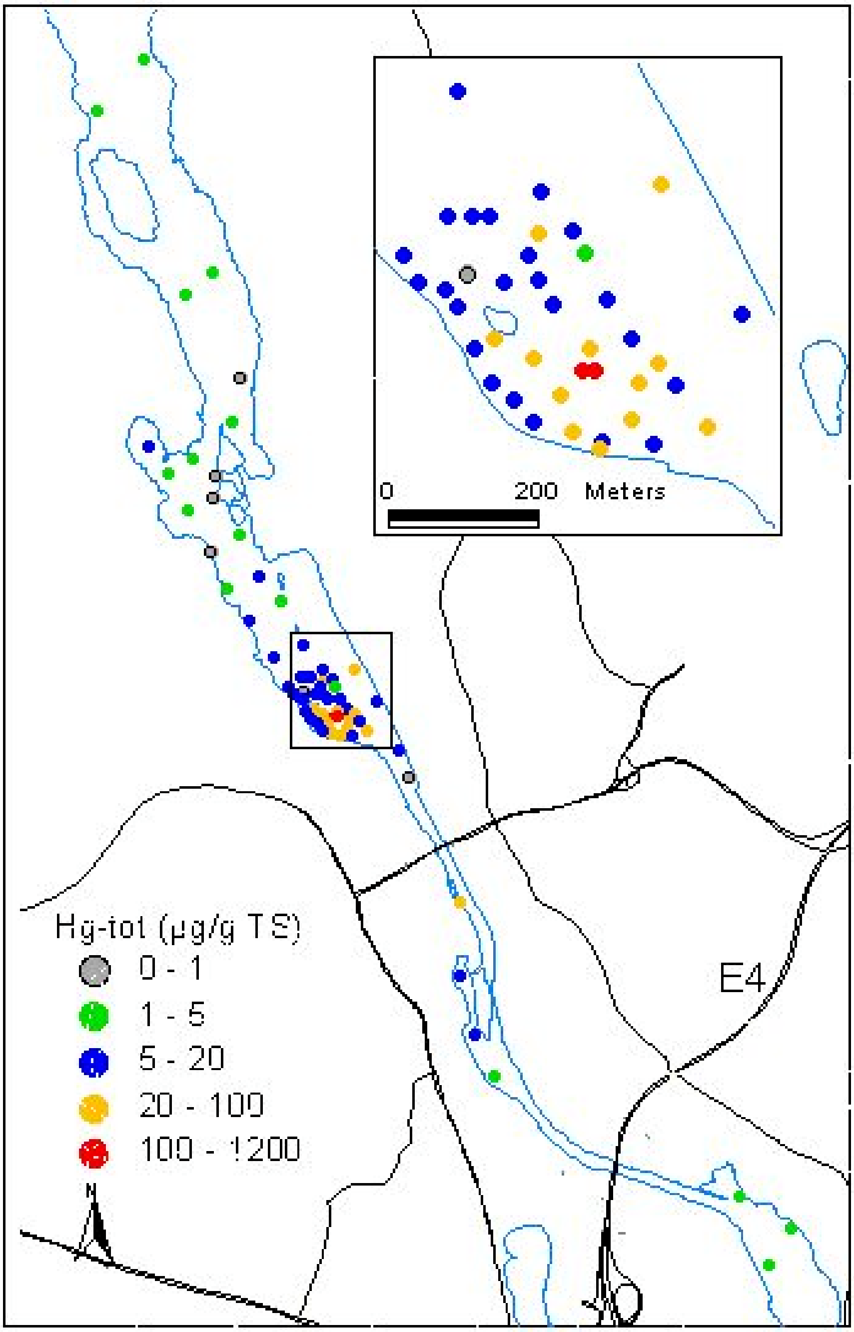

Figure 3. Concentrations of total-Hg in surface sediments (0-3; 0-5 cm). The Snäckviken area is enlarged in the inserted box.

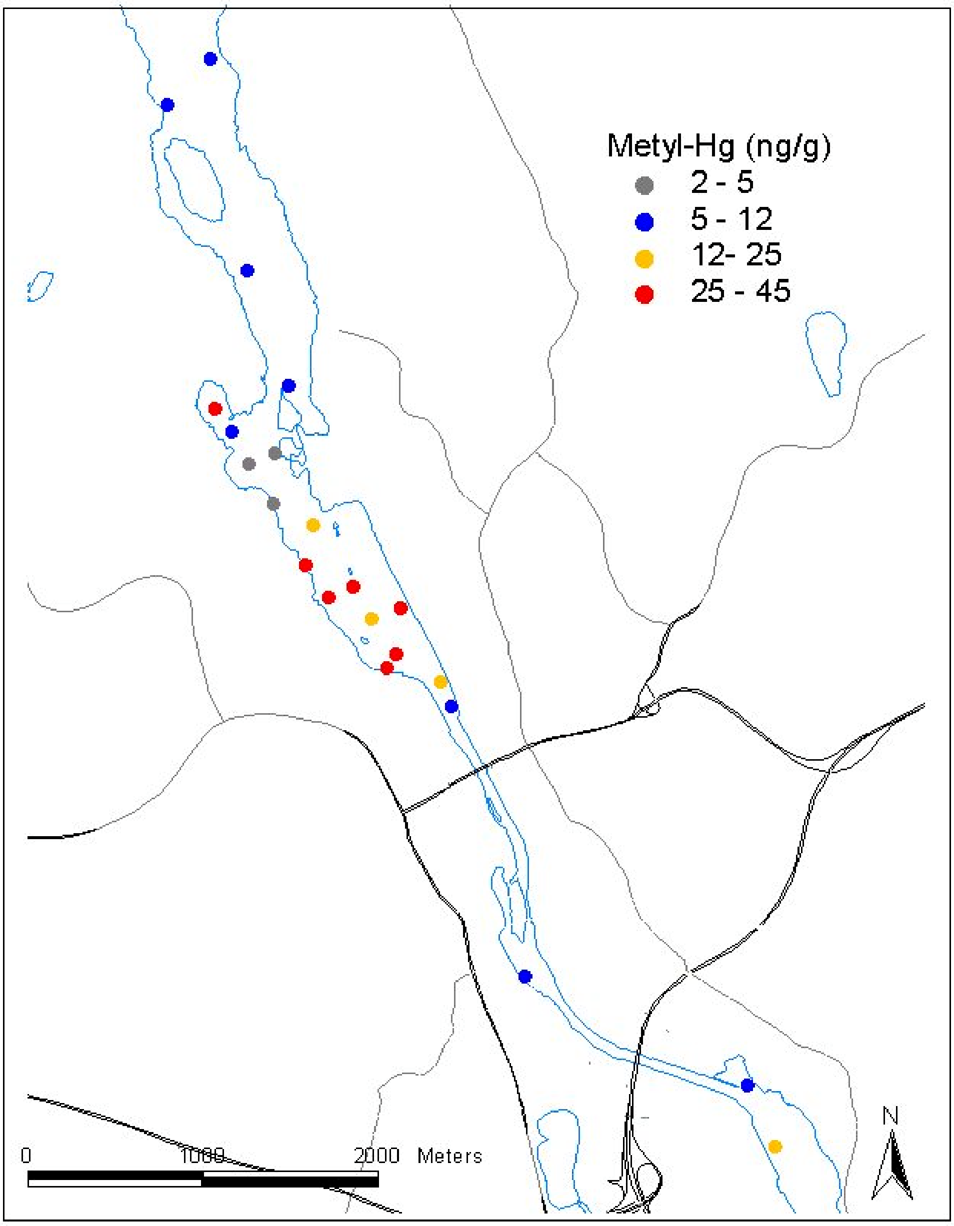

Figure 4. Methyl-Hg in surface sediments (0-3, 0-5 cm).

2.1.2 Other pollutants

Sediments from several stations in Lake Mälaren and the Baltic Sea were analysed for metals and certain organic pollutants (Viktor, 2000a). Selected results are presented in Table 2. At Krutholmen and Linasundet, concentrations of Cu, Pb and Zn were low to moderately high, according to Swedish EPA (1999a). Similar results also applied to sediments that were to be dredged and also included Cd and Ni (Parkman et al., 1998).

Concentrations of PCB (Polychlorinated Biphenyls) and PAH (Polycyclic Aromatic-Hydrocarbons) are very high at Krutholmen and those of HCB (HexaChloro-Benzene)are moderately high. The average concentrations of Σ7PCB in surface sediments in the centre of Stockholm (Östlund et al., 1998) is 45 µg/(kg 1% C), very similar to the levels found here (Table 2). Σ11PAH were on the average slightly higher in Stockholm compared to the present values. Concentrations of PCP (PentaChloro-Phenol) were regarded as low. Visual observations and smell also indicate that sediments at certain stations in Snäckviken (Krutholmen) are strongly contaminated with petroleum compounds. Their distribution appears very patchy but shows some coincidence with Hg.

Table 2. Concentrations of certain organic pollutants in sediments and the Swedish classification criteria (Swedish EPA 1999b). Concentrations in µg/kg dw and normalised to 1% organic C

	Linasundet	Krutholmen	Halls holme	High (Sw. EPA)	Very high (Sw. EPA)
Σ7PCB	13	45	40	4-15	> 15
HCB	0.05	0.11	0.19	0.2-1	> 1
Σ11 PAH	2078	2836	1165	800-2500	> 2500

2.2 Water column

Analyses of total-Hg and methyl-Hg in water at different locations in the area were conducted before, during and up to a year after the dredgings took place in the Snäckviken area 1996-97. Surface water was sampled in the whole area during one day. The amounts of Hg and methyl-Hg represent the conditions momentarily during one specific day. The total-Hg concentrations at the reference location near Kiholmen showed a mean of 3.4 ng/l ranging from 1.8-4.9 ng/l during the sampling period. General background concentrations for southern Sweden is 4 ng/l (Swedish EPA, 1999a). The methyl-Hg fraction was below detection limit (0.06 ng/l) and a maximum of 34% of total-Hg was present after a 0.45 µm filtration. The dredging operations caused no increase in the Hg concentrations in water at the reference site.

Water sampled at one location in the middle of Linasundet showed an average total-Hg concentration of 8.9 ng/l (range 4.7-13.9). The methyl-Hg fraction were also at this location below detection limit. After a 0.45µm filtration a maximum of 38% of total-Hg could be recovered. No relation of increased Hg concentrations and dredging operations could be established although concentrations in suspended solids were elevated. However, such a relation can neither be excluded since the sampling frequency was fairly low.

Both surface and bottomwater was sampled in Snäckviken. The surface water contained an average of 10.3 ng/l (range 3.9-17.2) of total-Hg. During the dredgings, a maximum of 35 ng/l was measured in surface samples. The methyl-Hg fraction increased from below the detection limit to a maximum of 0.12 ng/l (≈ 3 % of total-Hg) during the dredging period. The highest concentrations of total-Hg was found in bottom water in Snäckviken. The average total-Hg concentration was 28 ng/l (range 5.0-55) and increased to a maximum of 151 ng/l during dredgings. The methyl-Hg fraction increased to a maximum of 0.29 ng/l (0.4% of total-Hg) during dredgings. The methyl-Hg fraction was below detection limits after a 0.45 µm filtration and a maximum of 18% of the total-Hg was recovered after filtration.

2.3 Biota

Data on Hg in biota will be presented in this chapter. Concentrations of Hg in biota are regulated by a number of factors, e.g. ambient concentrations of Hg^{2+} and methyl-Hg, pH, salinity and feeding behaviour. In general, however, concentrations increase with trophic level because methyl-Hg biomagnifies (e.g., Pelletier, 1995). High concentrations at the base of the food-chain may thus indicate a risk for adverse effects occurring at higher trophic levels.

2.3.1 Zoobenthos

During 1996, zoobenthos were sampled at Linasundet and Kiholmen and analysed for methyl-Hg. In 2000, samples were taken in Linasundet, Kiholmen and Snäckviken. Data are shown in figure 5 and are generally lower than in the Baltic Sea stations (data not shown, see Parkman et al., 1998). Comparison with the methyl-Hg content in *Asellus* from a reference area and a Hg contaminated area in Småland (Hultsfreds kommun, 1998), and with two mesotrophic lakes (Parkman and Meili, 1993), suggests that concentrations in Snäckviken, but not in Linasundet, are slightly elevated. In the contaminated lake, sedimentary Hg was ca 1 µg/g, slightly lower than in Linasundet and Kiholmen.

Methyl-Hg in *Lymnea* shows that interannual variations can be significant, possibly due to different age/size of the specimen, but suggest that concentrations in both Snäckviken and Linasundet are elevated compared to the regional background. However, these elevations are very moderate as compared to the large differences in sedimentary Hg concentrations. Methyl-Hg in *Gammarus* show little variation in the area.

In these types of organisms, methyl-Hg is roughly 80% of total-Hg why maximum total-Hg should be ca 700 ng/g. Typical concentrations of total-Hg in macroinvertebrates from Swedish headwater lakes range from 50 to 500 ng/g dw

(Lindqvist et al., 1991). Compared to the reference lake in Småland, Hg in chironomids are slightly elevated but in *Asellus* they are not.

It is important to point out the different feeding strategies for the zoobenthos groups sampled in the littoral zone in Snäckviken. *Chironomus* and *Asellus* are detritius eaters feeding mainly on the sediment surface. *Lymnea* is a grazer and favours different kinds of attached algae. *Gammarus* is in some way opportunistic in its feeding habits. Depending on the circumstances *Gammarus* can either be a detrivore and feed on sediments or a predator with especially an affection for small *crustacean* i.e *Daphnids.* One other crucial issue when comparing the Hg concentrations in zoobenthos is the exposure time. *Lymnea* can be several years old and the age of large specimens of *Asellus* and *Gammarus* can also be counted in years. The reproduction of *Chironomus* is rapid and several generations of larvae develop to midgets during, e.g., a summer, so the exposure period is always much shorter compared to other zoobenthos.

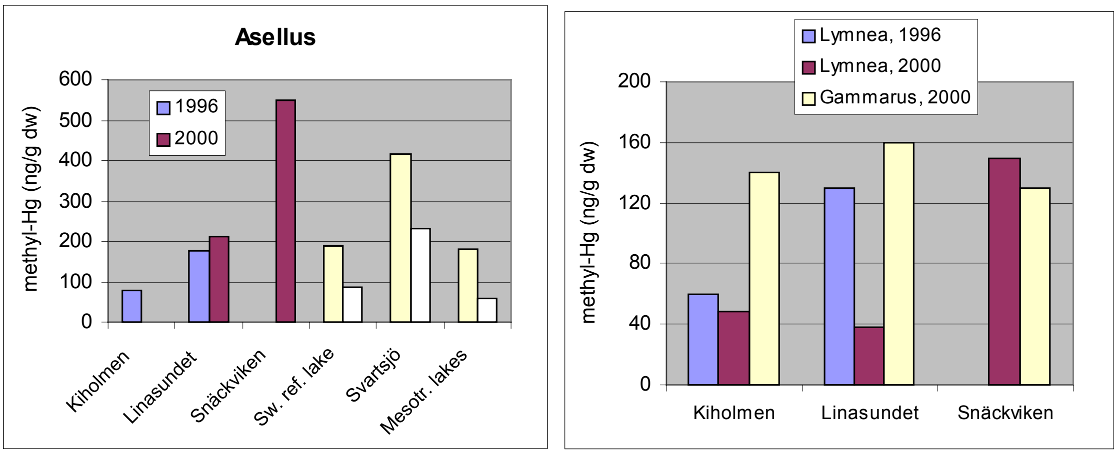

Figure 5. Concentrations of methyl-Hg in three species of zoobenthos. For *Asellus* comparison is made with two stations from Svartsjöarna (a reference lake and a Hg contaminated lake) and with two mesotrophic Swedish lakes, for which the two bars represent max and min values, respectively.

2.3.2 Zooplankton

Zooplankton display methyl-Hg in the same range as zoobenthos, as is usually the case (Lindqvist et al., 1991), and vary with nearly a factor 10 depending on the season (figure 6). Methyl-Hg in zooplankton from a Hg contaminated area in Småland also show a great variability depending on the season (Hultsfreds kommun, 1998). Data are not available to assess whether dredging leads to increased concentration in plankton.

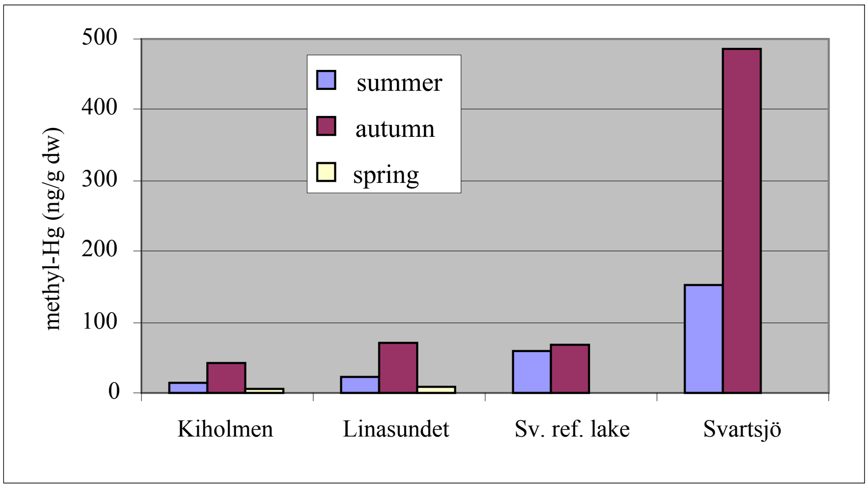

Figure 6. Methyl-Hg in zooplankton in two stations from Lake Mälaren, and two stations from Svartsjöarna (one reference lake and one Hg contaminated lake).

2.3.3 Fish

The concentrations of Hg in fish are higher than in lower organisms, which is usually the case because Hg biomagnifies. We have chosen to investigate transfer of Hg to fish by studying 1+ year old perch (*Perca fluviatilis*), because they are stationary and thus reflect the local conditions. Total-Hg in 1+ year old perch muscle is shown in relation to a major dredging operation in figure 7. Total-Hg in fish is mainly methyl-Hg. Kiholmen is a reference station in Lake Mälaren. It is obvious that Hg in young perch increases during dredging. Young perch feeds mainly on plankton. The concentration in 1+ year old perch during dredging corresponds to ca 0.7 mg/kg on wet weight basis (3.5 mg/kg dw), which must be considered very high for such young fishes.

Four- to five-year old perch from other sites in Lake Mälaren contain Hg levels of 0.15-0.18 mg/kg fw, i.e. ca 0.75-0.9 mg/kg dw (Lindeström, 2001). The database for national environmental monitoring of Hg in fish (www.ivl.se) contains no other data on Hg in young perch from Lake Mälaren or other nearby reference stations. However, in 15 background lakes studied during 1999-2002, levels in 1+ year old perch ranged from 0.08-0.45 mg/kg fw, that is, ca 0.4-2 mg/kg dw. For eight Swedish headwater lakes, Hg in small perch ranged from 0.2 to 2 mg/kg dw (Lindqvist et al., 1991). The background levels of total-Hg in 1+ year old perch in a reference lake in Småland was 0.12 mg/kg ww (0.62 mg/kg dw) (Hultsfreds kommun, 1998). The same study showed that in a lake with mercury contaminated fibre banks, a total-Hg content of 0.22 mg/kg ww (1.1 mg/kg dw) was found in 1+ year old perch. Thus, Hg concentrations in young perch from Södertälje channel are considered normal.

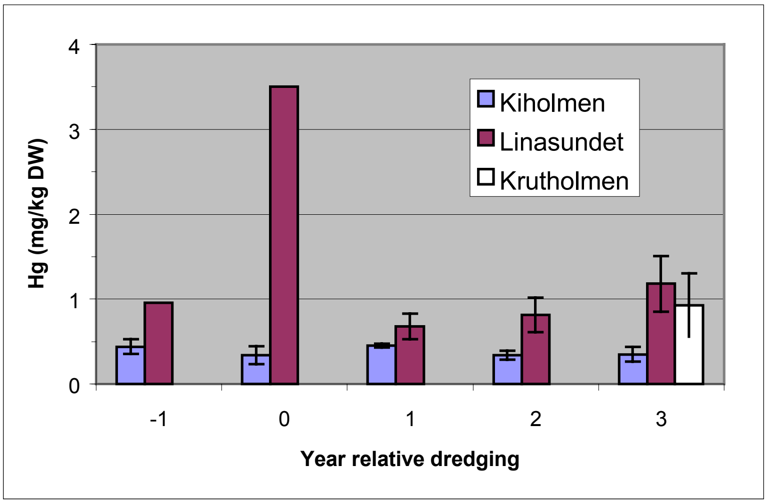

Figure 7. Concentrations of total-Hg in 1+ year old perch from three stations in Lake Mälaren before, during and after a dredging operation. Error bars represent standard deviation (n=6-10). Concentrations based on wet-weight is 5 times lower.

2.4 Main conclusions

Sediments show extensive contamination with respect to Hg, with highest concentrations generally occurring in Snäckviken. Chemical leachings indicate that bioavailability is low.

Water column concentrations of total-Hg in Snäckviken are only ca 2-3 times higher than regional background values, but increase during dredging.

The relatively low Hg concentrations in biota, as compared to concentrations in sediments, support the low bioavailability as indicated by chemical leachings. Similar concentrations in biota as found here are common in many Swedish ecosystems with much lower sedimentary concentrations of Hg.

3 Exposure

The major natural exposure pathways of sedimentary Hg are outlined in figure 8. The importance of each of these as well as possible anthropogenic influences will be assessed in this chapter. Other sources of Hg are also shown. Riverine input of Hg occurs by transport of Hg from upstream and by runoff from the drainage area. The regional background atmospheric deposition of Hg is ca 6 µg/m^2/yr (Kindbom et al., 1998), and is probably insignificant compared to the riverine input.

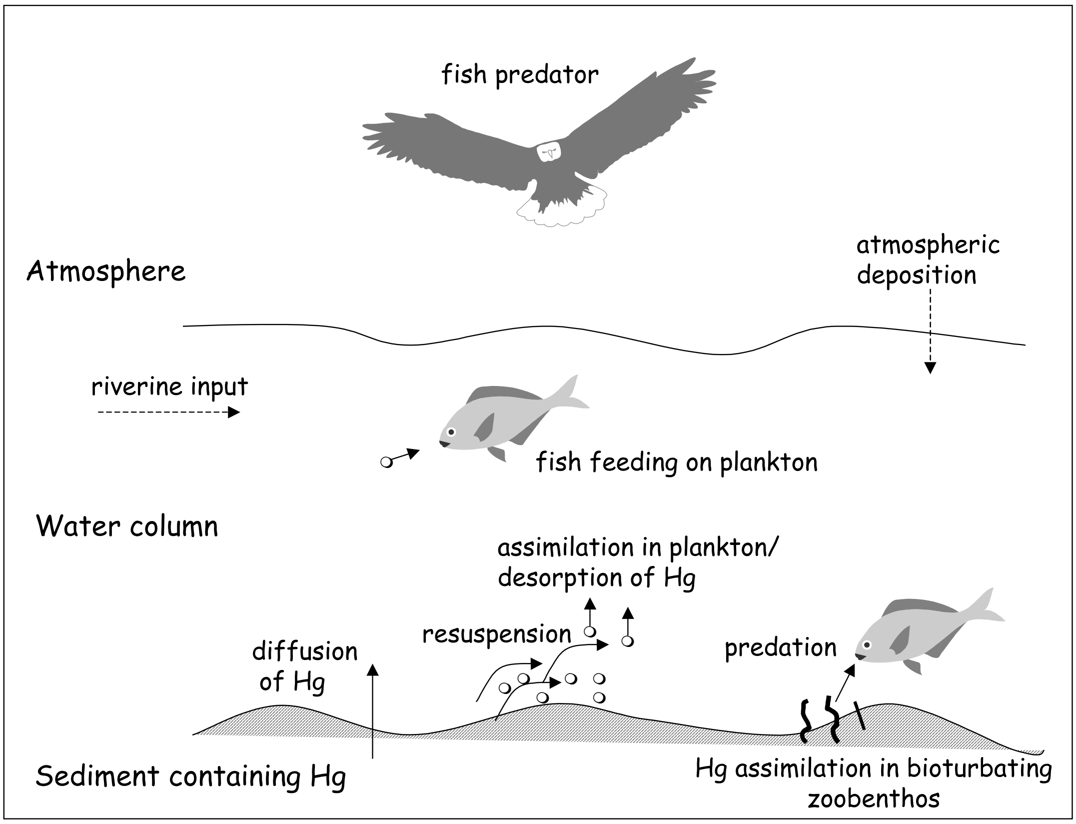

Figure 8. Schematic description of natural exposure pathways of sedimentary Hg to biota. Additional sources of Hg are marked with dashed arrows.

3.1 Diffusion

Diffusion is a physical process that may transport soluble Hg species from the pore-water into the water column, where they are available for uptake in biota. Mercury is present as two principal species in porewaters: Hg^{2+} and methyl-Hg. Diffusion relies on Hg concentrations in porewater being higher than those in the bottom water. None of the studies in Södertälje channel measured Hg in the porewater but the role of diffusion can still be estimated based on literature data.

A major chemical change between bottomwaters and sediments is that sediments frequently are anoxic and reducing. This affects both chemical reactions and microbial activity. As a consequence, concentrations of dissolved Hg^{2+} and methyl-Hg commonly increase in porewaters relative to bottomwaters (Gobeil and Cossa, 1993; Gagnon et al., 1996). Nevertheless, several studies indicate that Hg is fairly immobile in sediments (e.g., Gobeil and Cossa, 1993; Fitzgerald et al., 1997; Gill et al., 1999), because the amounts present in porewaters are such a minor fraction of total Hg.

Very few studies have attempted to actually measure the flux of Hg out of sediments. In shallow estuarine sediments, the flux of Hg^{2+} out of sediments was low and roughly comparable in magnitude to atmospheric deposition of Hg, i.e. ca 1-10 ng/m^2/d (Gill et al., 1999). This low mobility is explained by strong adsorption of Hg^{2+} at the sediment surface. Methyl-Hg adsorbs less strongly and is generally more mobile, in particular under reducing conditions (Gagnon et al., 1996). In the estuarine sediments, an average diurnal flux of methyl-Hg of 150 ng/m^2/day was measured, with highest fluxes during the dark period (Gill et al., 1999).

A few laboratory incubations of Södertälje channel sediments from the Baltic area showed fluxes of total-Hg between 170 and 1000 ng/m^2/day, with one extreme value of 7000 ng/m^2/day (Olsson et al., 1999). The bottom water in Snäckviken are likely less reducing than in the Baltic region, and Hg diffusion fluxes may therefore be lower in Snäckviken. Fairly low Hg concentrations in the water column in Snäckviken support this assumption. The escape of Hg due to sediment burial is at least 25-30$\times10^3$ ng/m^2/day and frequently > 100$\times10^3$ ng/m^2/day in Snäckviken. Thus, diffusion is unlikely to account for major losses of Hg from sediments to water. As discussed above, diffusion of methyl-Hg is expected to be more rapid than of total-Hg. In the water in Snäckviken and Linasundet, methyl-Hg was generally below 0.06 ng/l, i.e. less than 3% of total-Hg in water (chapter 4.2.) and was never measurable in the dissolved, i.e. diffusible, fraction. In a study of some Finnish lakes, sediments did not contribute methyl-Hg to the water column (Verta and Matilainen, 1995). Conclusively, diffusion of total-Hg and methyl-Hg from sediments in the Lake Mälaren region of Södertälje channel is likely of low importance, in agreement with measured levels of Hg in bottom waters.

3.2 Trophic transfer

The strongly bioaccumulative properties of Hg (in particular methyl-Hg) may lead to accumulation of Hg in sediment fauna. Predation by higher organisms on this fauna may transfer Hg up in the food-chain where concentrations may occur, i.e. biomagnification. The importance of this process is dependent on which benthic species that are present, their abundance and Hg concentrations, and to what extent higher organisms such as fish feeds on them. Several studies have suggested that this is an important pathway for Hg, from sediments to fish via intermediate organisms (e.g. Gagnon et al., 1996; Lawrence et al., 1999).

To investigate biomagnification trends, total Hg should be compared. Methyl-Hg in zooplankton averaged 50% (Parkman et al., 1998). In *Assellus* methyl-Hg constituted 80% of total-Hg (Viktor, 2000b). As a conservative estimate we assume that 70% is appropriate for all zoobenthos species. Total-Hg in different trophic levels are shown in figure 9, indicating that biomagnification occurs at all stations investigated in Lake Mälaren. As discussed in chapter 4.3, concentrations do not generally deviate from what could be considered normal in the Swedish environment. A stronger degree of biomagnification could, in theory, occur for fish that feed exclusively on benthic detrivores, e.g. *Asellus*. Those fishes, e.g. Bream (*Abramis abramis*) are not common in the region and transfer of sedimentary Hg to fish in the area should therefore not pose an ecological risk.

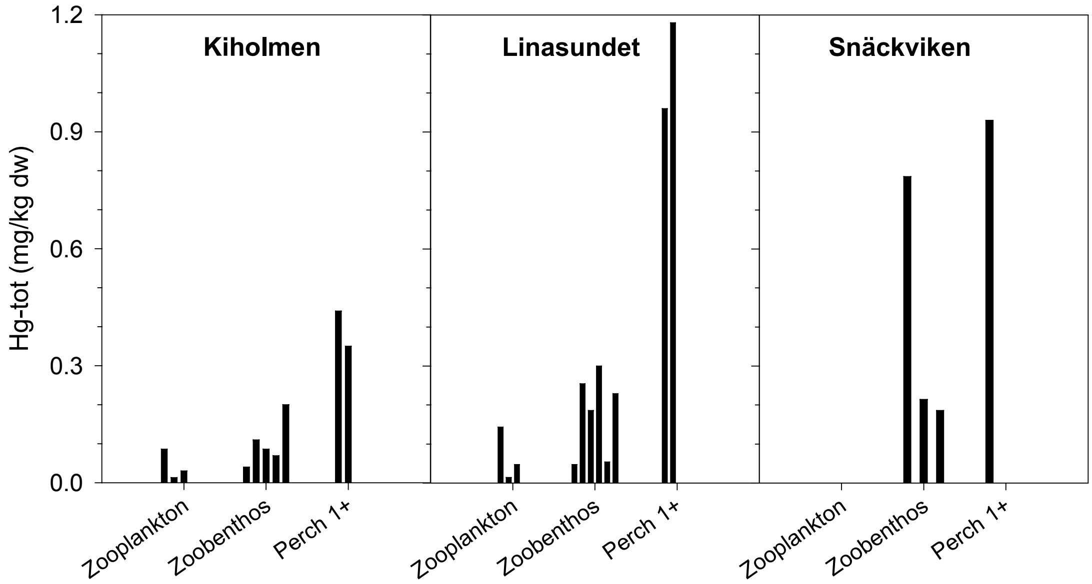

Figure 9. Biomagnification trends of Hg at the three stations during non-dredging periods. For zooplankton and zoobenthos methyl-Hg has been converted to total-Hg using appropriate con-version factors. The individual bars represent different seasons (plankton) and different species (benthos). Perch is represented by the average value of several specimen.

3.3 Resuspension

Resuspension is the process by which currents erode surface sediments so as to relocate sediment particles. This process has two consequences for sediment contaminants: (1) Contaminants are dispersed in the water column, becoming potentially available to biota, and (2) Contaminants become spread over a larger area. Mercury is strongly bound to the particulate fraction in natural waters and this is also the case in Södertälje (chapter 4.2). Consequently, resuspension has been identified as the major dispersion mechanism for sedimentary Hg in rivers (e.g., Hurley et al., 1998). Because Hg appears to show low bioavailability in these sediments (chapter 4.1.1), resuspended particulate Hg is not necessarily taken up by biota.

The variation in water flow through Södertälje channel over the year is shown in figure 10. With an approximate cross section area in Snäckviken of 3000-4000 m^2, the average net flow velocity will not exceed 1-2 mm/s. This is at least two orders of magnitude lower than the velocities required for eroding fine-grained sediments (ch.12 in Chorley et al., 1984). Therefore, erosion caused by the currents should be restricted to the most shallow, shoreward areas, where storm induced currents occasionally may erode sediments. Grain-size analyses did also show a relatively high content of fine particles (Sternbeck et al., 2000), which demonstrate the predominance of accumulation bottoms.

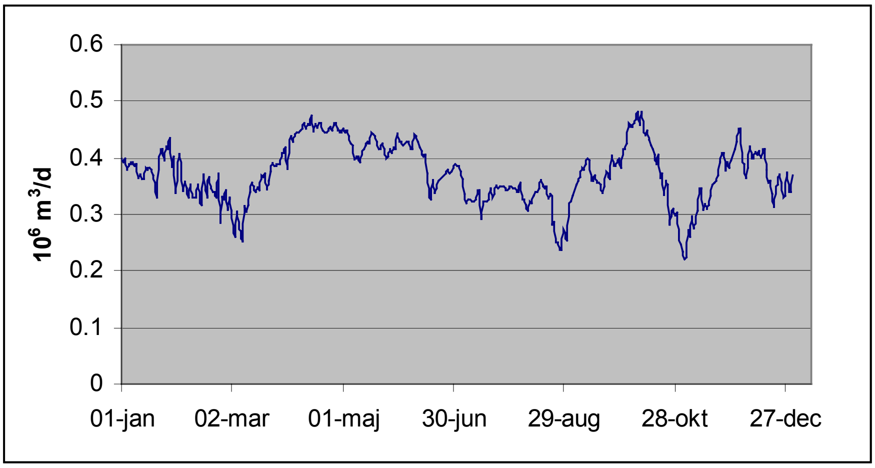

Figure 10. Daily waterflow through the culvert in central Södertälje. This represents more than 90% of the water passing Södertälje channel from Lake Mälaren.

Nevertheless, Hg concentrations are strongly elevated over a large area (figure 3) indicating extensive dispersion. This pattern persists at larger sediment depths. A certain relationship exists between the amounts of Hg in sediments and water depth. In figure 11 the amounts accumulated to 30 cm sediment depth are plotted versus water depth. Data are from Snäckviken and ca 500 m north. It can be assumed that emissions mainly took place close to the shore, where the highest concentrations are found (figure 3).

Water depth in that area is 7-8 m and explains the two highest values in figure 11. Except for these two stations, there is a fair correlation indicating that large amounts of Hg have been relocated to deeper areas in the channel, by means of erosion and resuspension.

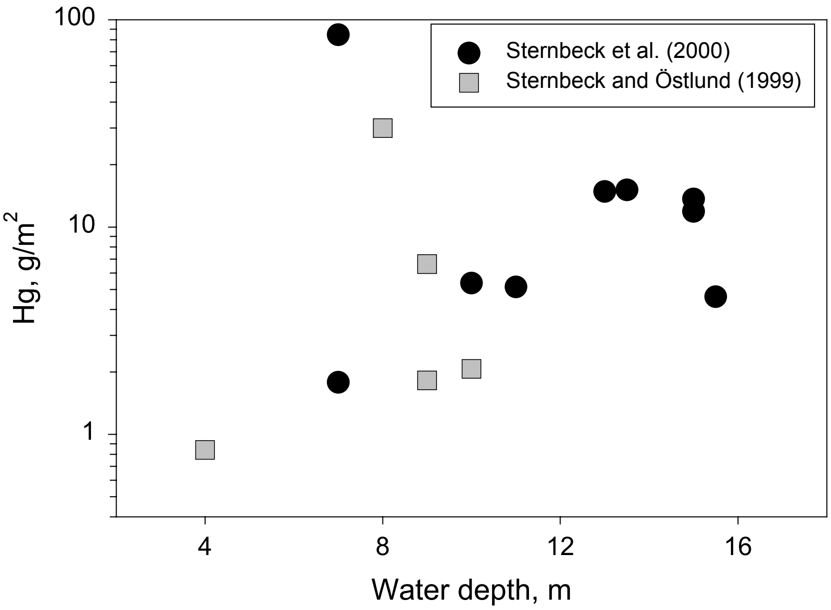

Figure 11. Relationship between the Hg inventory (g Hg/m^2) to 30 cm depth and water depth. Stations represent Snäckviken and nearby.

Radiochemical data also show that surface sediments are mixed to 2-10 cm depth (Sternbeck and Östlund, 1999). The variation of total-Hg over time is shown in figure 12. At Snäckviken, Hg levels have declined rapidly since the 1960ies, whereas sediments to the east of the shipping route (station 6) show evidence of recent supply of Hg. Their occurrence of phenyl-Hg at that site, suggests that this Hg derive from Snäckviken (Sternbeck and Östlund, 1999).

All these facts contrast with the waterflow being too low for causing widespread sediment erosion. Several studies have shown that large vessels are a major cause of sediment dispersion in coastal environments (Kelderman et al., 1998; Michelsen et al., 1998). Because an average of 12 large vessels pass Södertälje channel each day, this is most likely the explanation for the extensive dispersion of sedimentary Hg in the area from Snäckviken to Linasundet.

If jet boats will be considered to traffic the channel in the future, the hydrological effects of these vessels should be evaluated thoroughly in order to assure that resuspension will not increase.

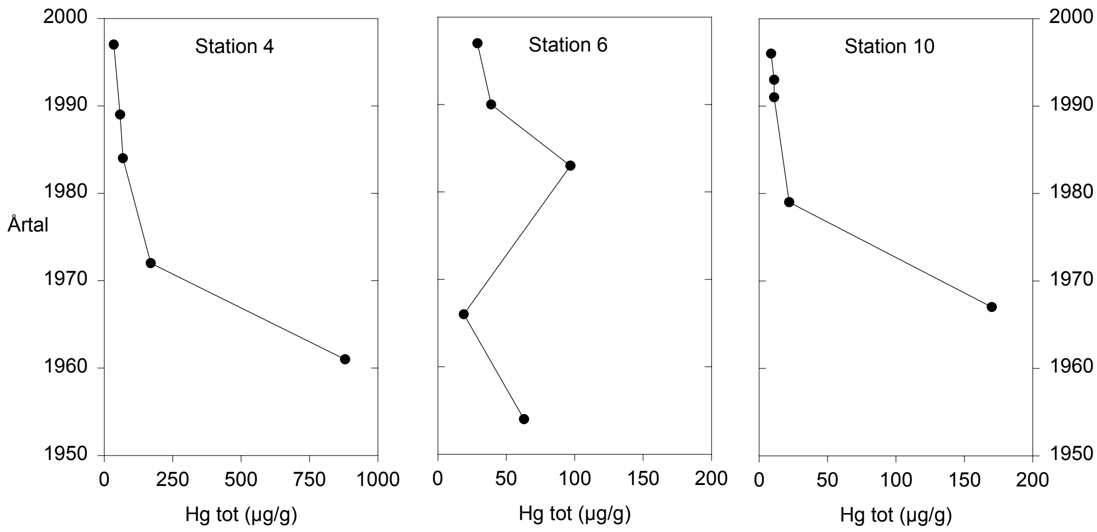

Figure 12. Variation in total-Hg over time at three stations. Station 4 is located south of Krutholmen, station 10 is located NNW and station 6 east of Krutholmen. Observe that each point represent ca 3-5 years.

3.4 Dredging

The risk for dispersion of Hg through resuspension should be compared with dispersion during dredging. It has been observed that Hg in the water column and in fish increase by orders of magnitude during dredging (Parkman et al., 1998).

Sediment traps were placed at various locations in the Södertälje area to sample resuspended materials around the dredging sites. Material that settled in the traps were analysed for total amount, and the organic- and metal content. A sediment trap located ca. 500 m north of Krutholmen showed that the lowest settling flux was 7.7 g/m^2/d and the highest was 41 g/m^2/d, occurring during the most intensive dredging period. The highest total-Hg content found in settling material was 40 µg/g and did also occur during dredging. Dredging leads to strongly increased Hg settling fluxes over a large area extending into the Baltic. During non-dredging periods, a Hg settling flux of ca 75 µg/m^2/d was measured, in rough agreement with the sediment burial rate of Hg (chapter 5.1).

3.5 Main conclusions

Diffusion is not likely to cause significant mobilisation of Hg from sediment to water column.

Sedimentary Hg may be transferred to fish via predation on zoobenthos, but the relatively low abundance of zoobenthos, and their moderate Hg content, suggests this to be a minor pathway in Snäckviken.

The widespread occurrence of high Hg levels in sediments demonstrates that resuspension must be an important dispersion mechanism. Most likely, the passing of large vessels causes resuspension. This must be considered if intensified vessel traffic is planned for the future.

Dredging leads to elevated concentrations of Hg in young fish and temporarily in water column, and to increased settling fluxes of Hg.

4 Effects

Mercury differs from most other metals by being strongly bioaccumulated and biomagnified in the aquatic food-chain, mainly due to the presence of the lipophilic methyl-Hg (see Pelletier, 1995). Bioaccumulation implies that chronic effects may be more important than acute effects. Biomagnification implies that effects at higher trophic levels are of concern, in particular accumulation in fish and subsequent uptake in man or other top predators. Whether biological effects of Hg occur *in situ* are difficult to assess for several reasons, e.g., (1) numerous biological stressors other than Hg are present and it is often difficult to link a certain effect to a certain stressor; (2) effects of Hg tend to occur in the top of the food-chain and are thus difficult to link to a certain restricted area; (3) variations at the community level are caused by numerous environmental factors other than pollutants.

This chapter reviews studies from Södertälje channel directed at identifying effects *in situ* or in laboratory by several approaches. 1) Measured concentrations in various media are compared to toxicity-based threshold values; 2) Bioassays of the sediments on four different organisms; 3) Variations in the benthic community structure that may be pollutant induced; 4) The natural production of perch roe.

4.1 Measured concentrations and toxicity-based threshold values

There are no Swedish guidevalues for Hg in surface waters. Hempel et al. (1995) performed six different bioassays on bacteria and nematodes of different mercury compounds. The most sensitive organisms appeared to be the nematodes with EC_{50}[1] values of 15 µg/l for methyl-Hg, whereas Hg^{2+} was orders of magnitude less toxic. The Canadian Water Quality Guidelines puts the limit for less serious concentrations of total-Hg at 26 ng/l and methyl-Hg at > 4 ng/l. Total-Hg exceeds the guideline during dredging, and occasionally also during non-dredging but then only in bottomwaters. In the upper water column, total-Hg was generally less than ca 15 ng/l and methyl-Hg less than 0.06 ng/l during non-dredging periods, i.e. lower than the Canadian guidelines.

Possible biological impact from sedimentary Hg is more difficult to assess because bioavailability of Hg differs widely between different types of sediments. Furthermore, toxicity based threshold values rarely differentiate between total-Hg and methyl-Hg. Methyl-Hg commonly constitute 0.1-1% of total-Hg in sediments but is a significant or

[1] EC50 is the concentration of a compound which causes 50% of a population to be adversely affected.

even major fraction of Hg in biota. The Swedish Environmental Quality Guidelines (Swedish EPA, 1999a) are not related to the risk for adverse effects and are thus not relevant in this context. The Canadian Sediment Quality Guidelines sets the level for probable effects (PEL) at 0.49 µg/g. This value is likely overprotective in most regions. For instance, it was shown that the *Chironomus* population of a Hg polluted lake started to be adversely affected above ca 40 µg/g (Suchanek et al., 1995). Sediments with Hg concentrations up to 170 µg/g showed no acute toxic effect to an amphipod (Horne et al., 1999). In a similar study, it was concluded that the site-specific threshold for total-Hg toxicity to amphipods exceeded 4.1 µg/g (Sferra et al., 1999). Conclusively, preliminary guidevalues suggest that a toxic effect of these sediments cannot be excluded, although several field studies contradict this suggestion. Bioassays have been performed and will be summarized in chapter 6.2.

Concentrations of methyl-Hg in zoobenthos and zooplankton (chapter 4.3) do not deviate from values normally encountered in Sweden. We are not aware of any threshold values for toxic effects of Hg in these organisms. Mercury in young perch is very high during dredging, but there appears to be no consensus regarding the level at which physiological effects may start to occur. During non-dredging periods, Hg in young perch is not elevated.

4.2 Bioassays

A multispecies test was carried out to survey the impact on aquatic organisms exposed to sediment from Södertälje area (Viktor, 2000a). Surface sediment samples representing 0-5 cm was collected from 9 locations, during spring 1998. Samples were collected from three locations in Lake Mälaren (see figure 1) and six locations in the Baltic area south of Södertälje. The primary aim of the study was to evaluate the biological impact from the area where the dredged sediments were dumped, a deep in Hallsfjärden. Since the study was focused on the Baltic area, all the tests were performed in a flow-through system containing brackish water. The organisms were representing major groups from the Baltic littoral zone (Table 3). Except for *Mysis*, however, the organisms are also common in freshwaters.

A summary of the results are presented in Table 4 together with corresponding Hg levels. Sediments from Halls holme show the highest toxicity, whereas Krutholmen does not appear more toxic than Linasundet or Kiholmen in spite of the high Hg-levels at Krutholmen. Conclusively, sediment toxicity tests do not correlate with Hg levels suggesting either that the observed effects may be caused by other pollutants, or that concentrations is a poor measure of bioavailability.

Table 3. Organisms used in sediment toxicity tests.

Group	Taxa	End-points
Molluscs	*Lymnea peregra*	Survival, reproduction, growth
Crustacean	*Mysis mixta*	survival, reproduction, growth
Amphibia	*Bufo bufo*	tadpole survival and growth
Pisces	*Perca fluviatilis*	hatchability and survival of embryo/larvae

Table 4. Overview of sediment toxicity studies. 0 no observed effect compared to the reference; + observed effect not statistically significant; ++ observed effect statistically significant ($p < 0.05$)

Method	Kiholmen	Linasundet	Krutholmen/ Snäckviken	Halls holme
Hg µg/g dw	1.5	1.9	27	6.2
Lymnea adult mortality	0	0	0	0
Lymnea reproduction	0	0	+	+
Lymnea young growth	++	0	++	0
Mysis adult mortality	+	+	+	++
Mysis young growth	0	0	0	+
Bufo tadpole mortality	0	+	+	++
Bufo tadpole growth	++	++	++	++
Perca embryo hatching	0	++	0	++
Perca larvae deformation	0	+	+	++

All the bioassay data were also evaluated against sedimentary pollutants using multivariate techniques (Appendix 1). Overall, this evaluation also show that sediments from the Baltic area (Halls holme) were more toxic than sediments from the Lake Mälaren area. Toxic effects were generally not related Hg or other metals but rather to organic pollutants such as PCB and PAH. The number of young *Lymnea* was related to Hg but also to certain PAH. There were no significant differences in toxicity between sediment Linasundet and Krutholmen, although pollutant concentrations are much higher at Krutholmen. Possibly, this indicates that bioavailability differs less than concentrations.

4.3 Benthic community structure

The benthic fauna structure in Snäckviken has been documented (R. Brown in Sternbeck et al., 2000). When interpreting the results, it must kept in mind that sampling took place in late fall which is not optimal for benthic fauna. Two common measures of the ecological state of the zoobenthos community are the Shannon-Wiener index and the O/C index (Oligochaete/Chironomid) (Swedish EPA, 1999a). The Shannon-Wiener index is a measure of both diversity and abundance. The SW-indexes are very low throughout most of the area (figure 14). In figure 13 the SW-index is plotted versus water depth and total-Hg, respectively, which indicates that the abundance and diversity of benthic fauna is related to water depth and not necessarily to Hg or other pollutants. Similar observations have been made elsewhere (e.g., Suchanek et al., 1995; Horne et al., 1999).

The O/C indexes show high abundance of oligochaetes compared to chironomids which are common in polluted areas. A correlation with all environmental variables showed that total solids (inversely related to sediment water content) was most influential on fauna abundance and diversity, followed by methyl-Hg and water depth (R. Brown, in Sternbeck et al., 2000).

A renewed sampling of zoobenthos for MeHg analysis was conducted during October 2000 (Viktor, 2000b). One of the main purposes was to examine if the zoobenthos community structure was different at the shore of the island of Krutholmen compared to deeper parts of Snäckviken. The collection of specimen for analysis was focused on taxa from the following groups: Molluscs (*Lymnea* sp) and Crustacean (*Mysis relicta* and *Asellus aquaticus*) . Sufficient amount of organisms for analysis, representing the three groups, were found at the shore of Krutholmen but not in the deeper parts of Snäckviken. Several other benthic organisms, not previously reported from the area, were found in the littoral zone of Krutholmen, e.g. insectslarvae representing family *Ephemeroptera, Limnophilus* and *Odonata* which are indicators of a good water quality. The presence of aquatic vegetation at the shore indicates that this area is less physically disturbed than the deeper regions, and that physical disturbance may be a significant factor in reducing the benthic community diversity in Snäckviken.

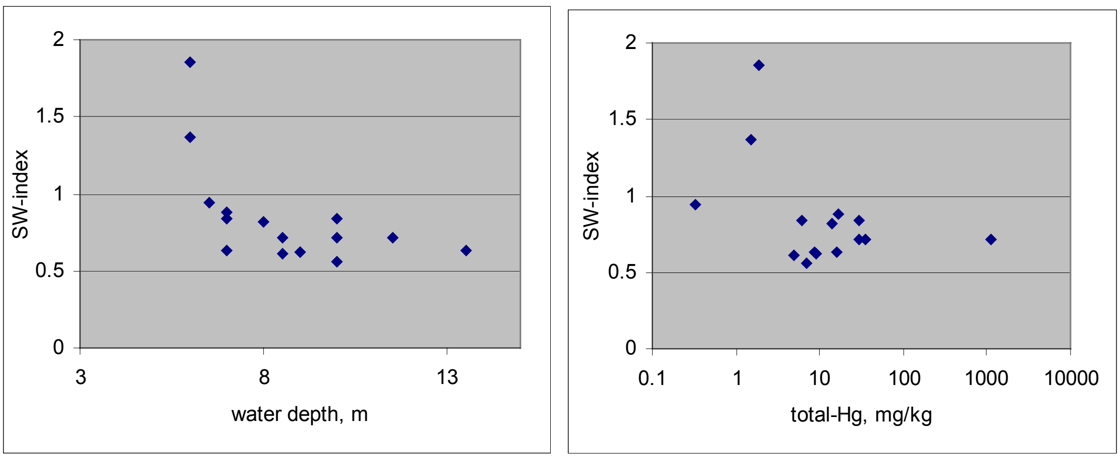

Figure 13. Relationship between water depth or total-Hg and quality of zoobenthos. A high value of the SW-index means good environmental quality.

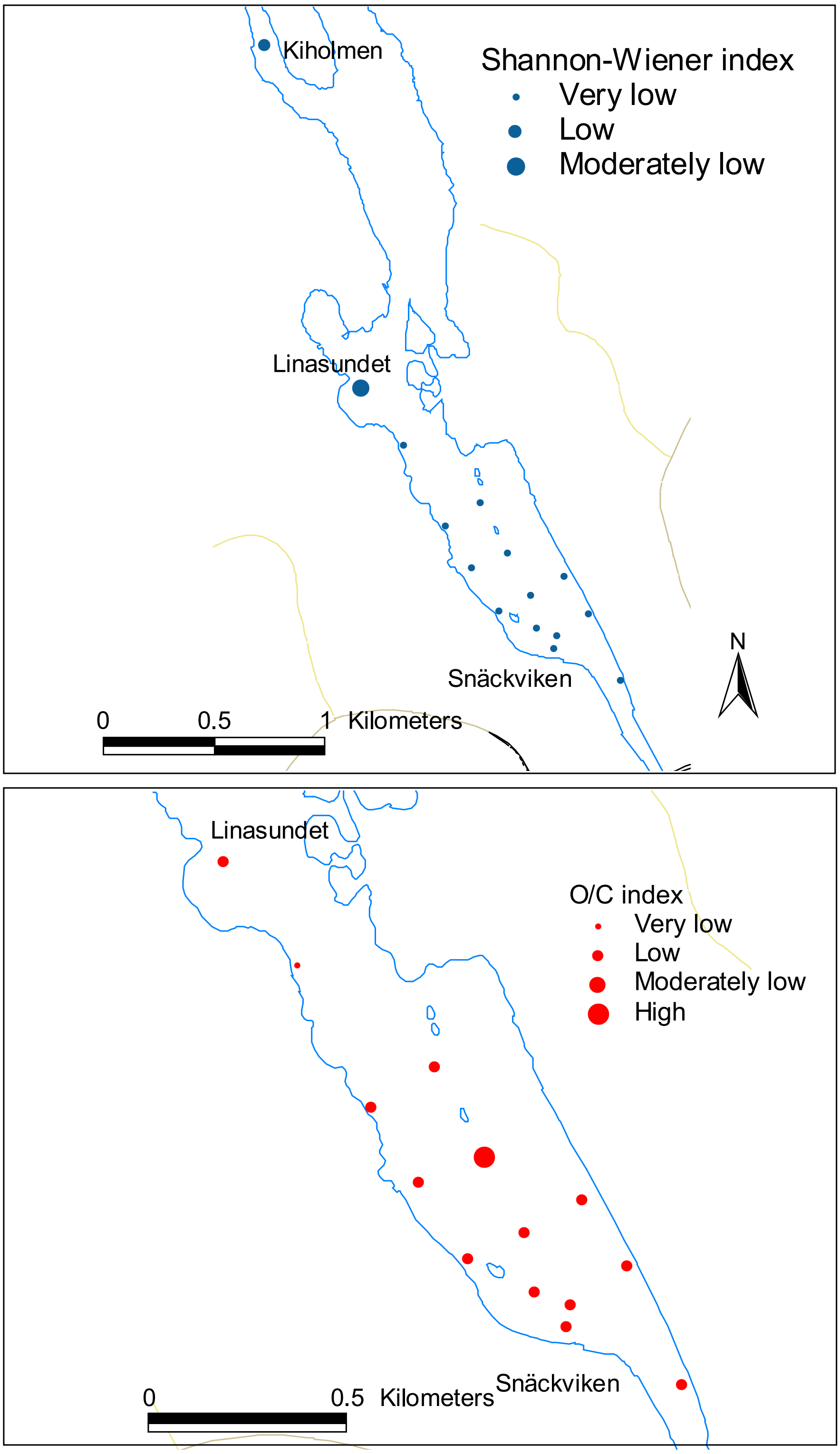

Figure 14. Indexes for benthic fauna. A high Shannon-Wiener index means high diversity/abundance. A high O/C index means that the environment is disturbed.

4.4 Natural reproduction of perch (*Perca fluviatilis*)

During spawning, female perch attach a single continuous string of roe with a length of ca. 1 m at depths normally less than 2 m on vegetation, tree branches etc. To estimate spawning occurrence, 1.5 m long spruce tops were used as artificial spawning substrates. The substrates were placed at four different suitable locations in Lake Mälaren and Hallsfjärden. In order to determine hatchability, survival and occurrence of deformation on naturally produced roe, samples from the strings were collected and transported to the laboratory.

Perch spawning occurred in Snäckviken at the northern shore of Krutholmen in spring 1998. Incubation of the roe in the laboratory in Lake Mälaren water was conducted during 20 days at 10°C, close to the normal temperature. The results are presented in Figure 15 and compared with roe collected in Hallsfjärden and at a reference site at the Baltic coast in northern Uppland.

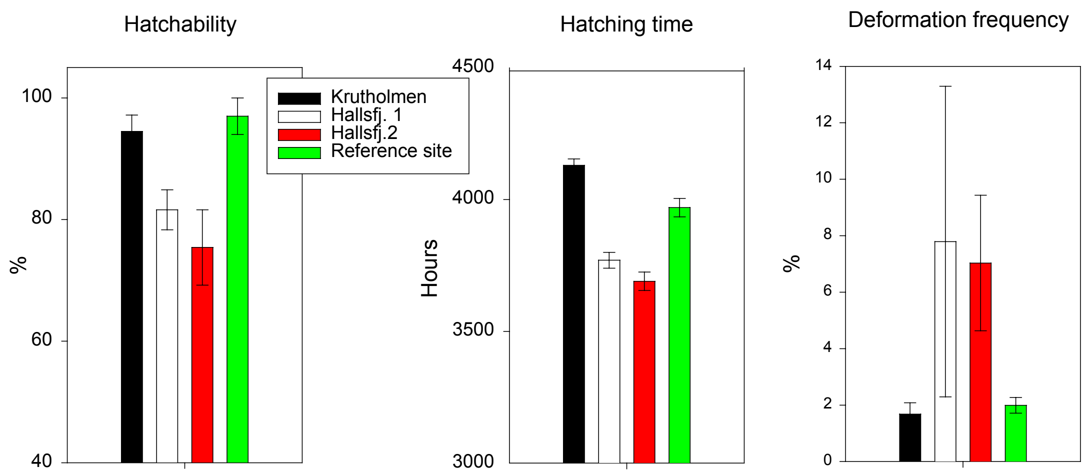

Figure 15. Embryo/larval development from naturally produced perchroe.

No developmental parameter for perch roe produced in Lake Mälaren differ from the reference values obtained for roe produced in the Baltic sea. A significant ($p<0.05$) decreased hatchability and hatching time were found for roe produced in Hallsfjärden. The effects measured show that the parental fish in Hallsfjärden are affected and produced a roe with poorer quality compared to e.g. Krutholmen. Deformations on hatched larvae observed in Lake Mälaren was comparable with the numbers found at the reference site and agree with the background levels found in low urbanised Baltic coast zones (Karås et al., 1991; Sandström et al., 1996). This study indicates that water quality is good at Snäckviken/Krutholmen, which is in agreement the low levels of Hg in the water column (chapter 6.1).

Fishsampling with gillnet, for tissue analyses, was conducted in the autumn 2000. When comparing the three different fishsampling stations in Lake Mälaren, it can be noted that the highest number of small perch (0+ year) was caught in the Snäckviken area, close to the island of Krutholmen. This strengthens the hypothesis that the perch reproduction is not adversely affected in the Snäckviken area.

4.5 Main conclusions

Sediments from Snäckviken, Linasundet and Kiholmen show low to intermediate toxicity compared to Hallsfjärden, and the toxicity is not correlated to Hg.

Perch successfully reproduce in Snäckviken and the embryo/larvae survival and status is comparable with reference sites.

The zoobenthic fauna showed very low indexes in Snäckviken but no positive correlation to Hg content in sediments could be established. Possible effects of Hg may be obscured by the presence of other pollutants such as PAH and PCB, and petroleum compounds.

5 Gaps in knowledge

The hydrology is poorly known and a detailed hydrological investigation would allow to identify regions with high risk for dispersion of Hg. In particular, the effect of vessels on resuspension should be studied in the field at locations with different water depths.

To assess whether Snäckviken contributes with Hg to Igelstaviken and Hallsfjärden, dispersion of Hg from Snäckviken could be quantified by measuring the flux of Hg through the culvert and through Linasundet.

If remediation is to be considered, the nature of bulk Hg in sediments should be determined to select proper cleaning techniques or disposal option.

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Appendix 1. Multivariate evaluation of the toxicity and reproduction-success in a multispecies testsystem in the presence of sediments from Södertälje.

Jenny Olsson and Tomas Viktor, IVL

Introduction

This report presents a multivariate evaluation of data from a multispecies test system. The measured response parameters include toxicity and reproduction success for aquatic organisms representing the Baltic littoral zone. The organisms are typical for a Baltic coastal area but they are widespread and also common in freshwater. The evaluation also consists of available data of metals and organic pollutants in sediments from eight different locations in the Södertälje region (Viktor, 2000). Three locations are situated in Lake Mälaren ; Kiholmen (freshwater reference), Linasundet (northern part) and Krutholmen (Snäckviken). The baltic area south of Södertälje are represented by four locations ; Halls holme (northern part of Hallsfjärden), Bränningeviken (western part of Hallsfjärden), Tipplatsen (dumping aera for dredged sediment, central part of Hallsfjärden) and Fläsklösa (southern part of Hallsfjärden). The brackish water reference site is located in Forsmark in northern Uppland.

Data

It is often difficult to extract valuable information from large data matrices consisting of a large amount of variables in a conventional way, for example by looking at trends. Multivariate methods is an effective statistical tool for evaluating large amounts of data. It makes it possible to detect correlations and relations between parameters by comparing all of them simultaneously. Multivariate methods makes it possible to conclude which parameters that have the main impact on a certain response parameter. When the number of observations/experiments is limited, as in this evaluation, the models are sensitive which has to be considered when interpreting the results.

Each effect variable has been scaled to express the toxicity. For example a high value of the animals weight do not correspond to a high weight but to a high toxic effect on the weight (thus a low weight increase). The parameters that not express toxicity originally were scaled based on the data for the eight locations. Originally the highest value for those variables expressed the least toxic effect. This location were from the scaling given the value 0 for toxicity. The values in the other locations were subtracted from the highest value to achieve an appropriate value of the toxicity.

Multivariate methods

Assume that three variables, X1, X2 and X3 can describe a system. Each observation (experiment) is represented by a position in the three-dimensional space based on its values of the three variables. Several experiments result in a swarm of positions. A vector using the least-squares method approximates the swarm. This vector is called the first principal component and describes the largest variance among the observations in the system. Another vector, perpendicular to the first, is also calculated. It describes the direction for the second largest variance.

The two calculated vectors will together form the slope of a two-dimensional plane in the three-dimensional space, see figure 1. The plane is fitted to minimize the sum of the quadrants on the distance from the objects to the plane. Thus the plane is the best fitted to the results of the experiments.

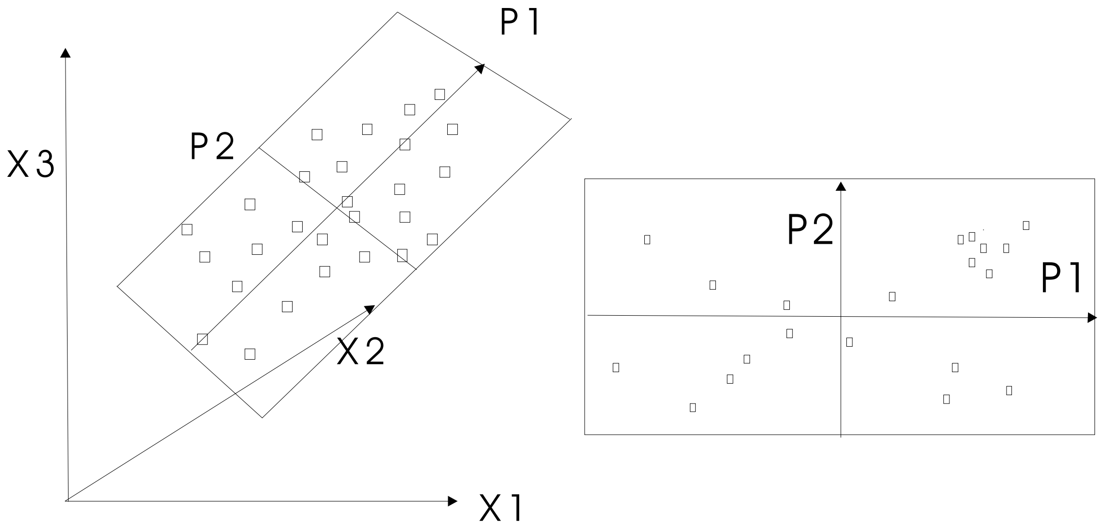

Figure 1. Principles for multivariate statistics.

The two principal components have reduced the dimensions from three to two. The swarm of observations is then projected on to the two-dimensional plane, se figure 1. This plane can be studied and the relations between different observations can be evaluated. The result of the calculations is that the system can be described in a two-dimensional space instead of a three-dimensional. Several principal components can be calculated but often three or four are sufficient to describe the major part of the variance of a system. A multi-dimensional space can also be summarized to a two-dimensional plane.

Principal Component Analysis (PCA)

The result from a PCA provides information about co-relations between parameters and observations and also some information about the relations between X- and Y-parameters. All parameters are considered as X-parameters (independent parameters) in a PCA. A PCA-model receives a percentage value showing how much of the variance in the X-parameters that is described by the model. It is desirable to achieve a percentage value as high as possible. It is possible to get a 100% explanation of the variance in the X-parameters with a sufficient number of principal components. However, it is not interesting to include too many components in the model because then the noise is included as well. The aim with a PCA is to achieve a high percentage value with as few principal components as possible. The theory behind PCA is described in Chatfield and Collins, 1992. The theory for PCY is identical to the theory for PCA. The difference is that in a PCY only Y-variables are included. Correlations between Y-parameters are studied.

Partial Least squares (PLS)

In a PLS it is studied how the X-parameters affect the Y parameters (output). A PLS connects the Y- and X-spaces. X-parameters of great importance to the Y-parameters are detected in a PLS-analysis. The background theory behind PLS-modelling and it applications, may be further studied in Geladi and Kowalski, 1985. The different biological parameters and organic pollutants used in the modelling are shown in table 1 and 2.

Table 1. Biological parameters measured in the multispecies system.

Variables	**Unit**	**Description**
P H1, P H2	%	*Perch* Hatched embryos of total examined eggs
P D1, P D2	%	*Perch* Deformed larvae of total hatched larvae
P ST	days	*Perch* Median survival time of embryo/larvae stage
B W	mg	*Bufo* Weight of tadpoles after exposure during 9 weeks
B L	mm	Lenght as above
M W	mg	*Mysis* Weight of adult after exposure during 9 weeks
M L	mm	Lenght as above
L M	%	*Lymnea* Mortality for adult
B M	%	*Bufo* Mortality of tadpoles
M M	%	*Mysis* Mortality for adult
L REP	number	*Lymnea* Number of eggclutches
M REP	number	*Mysis* Confirmed reproduction
L NO Y	number	*Lymnea* Number of young
L W Y	mg	*Lymnea* Weight of young
M NO Y	number	*Mysis* Number of young
M W Y	mg	*Mysis* Weight of young

Table 2. Chemical sediment parameters.

abbrevation	compound	unit
Cu	Copper	mg/kg dw
Hg	Mercury	mg/kg dw
Pb	Lead	mg/kg dw
Zn	Zink	mg/kg dw
PCB	Polychlorinated biphenyls	µg/kg dw
HCB	Hexachlorobenzene	µg/kg dw
Phen	Phenanthrene	µg/kg dw
Fla	Flouranthene	µg/kg dw
Pyr	Pyrene	µg/kg dw
Chrys	Chrysene	µg/kg dw
Ben(b)	Benzo (b) flouranthene	µg/kg dw
Ben(a)	Benzo (a) pyrene	µg/kg dw
Indo	Indeno(cd)anthracene	µg/kg dw
Ben(ghi)	Bens (g,h,I) perylene	µg/kg dw
PAH	Total amount of Polycyclic aromatic hydrocarbons	mg/kg dw
PAH c	Total amount of carcinogenic PAH's	mg/kg dw

Results

PCA

The model is composed of three Principal Components (PC) and explains a total of 73 % of the variance of all variables included in the matrix, both X (pollutants) and Y (effect variables). The score- and loading plots for the PCA-model are shown in figures 2 and 3. The score plot shows the place of the locations in relation to each other based on the data of all variables. In the corresponding way the loading plot shows the place of the variables in relation to each other based on data from all locations.

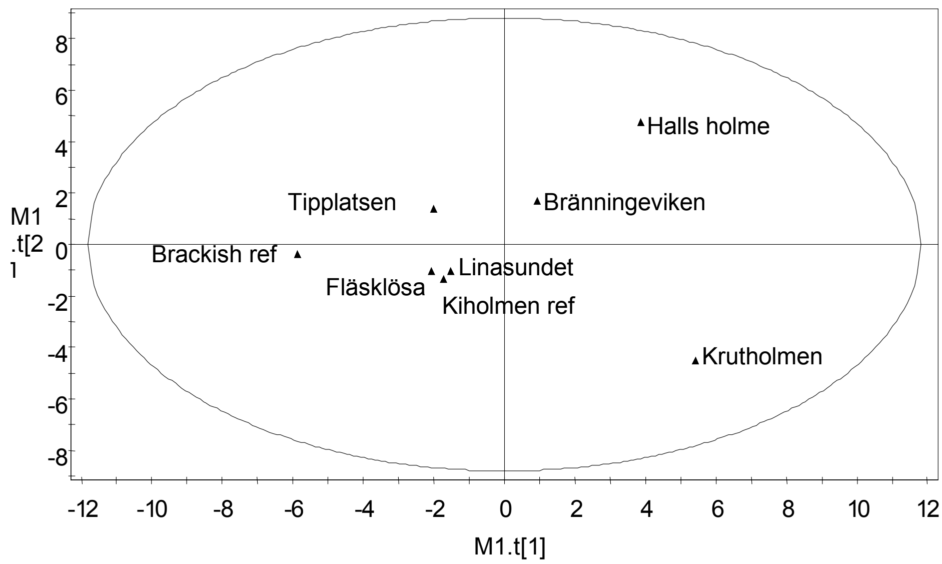

Figure 2. Score plot based on all parameters in the matrix

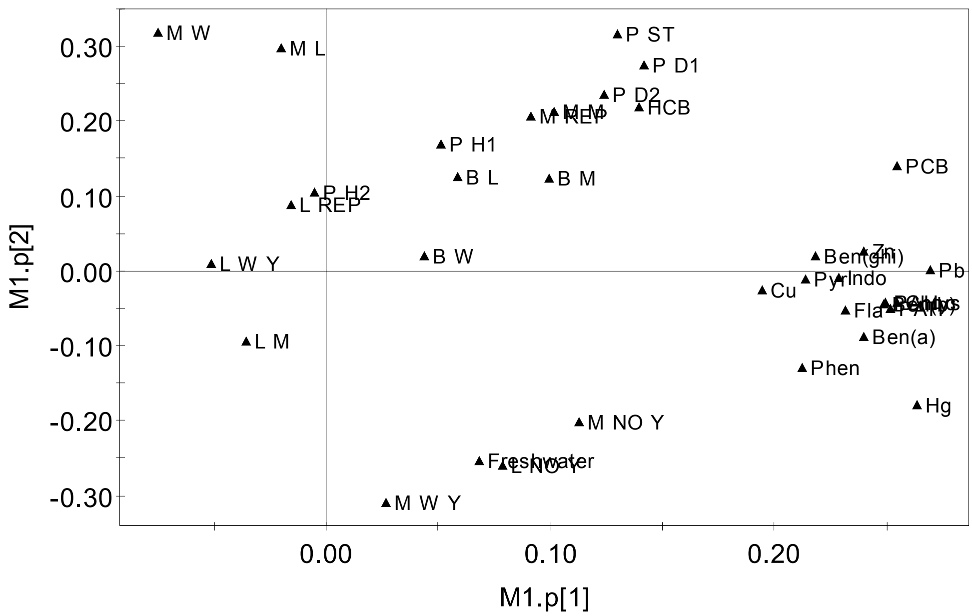

Figure 3. Loading projection showing the correlations between all parameters included in the matrix.

The values of analysed parameters at the locations Halls Holme (the most pronounced impact on the aquatic organisms of Baltic Sea locations), Kiholmen (freshwater-reference) and Linasundet are compared to Krutholmen. The results are shown in figures 4, 5 and 6. The measured effects on the different organisms and analysed pollutants in sediment from Halls holme, are more pronounced/higher values compared to Krutholmen, for the parameters with positive bars and lower values for the ones with negative bars.

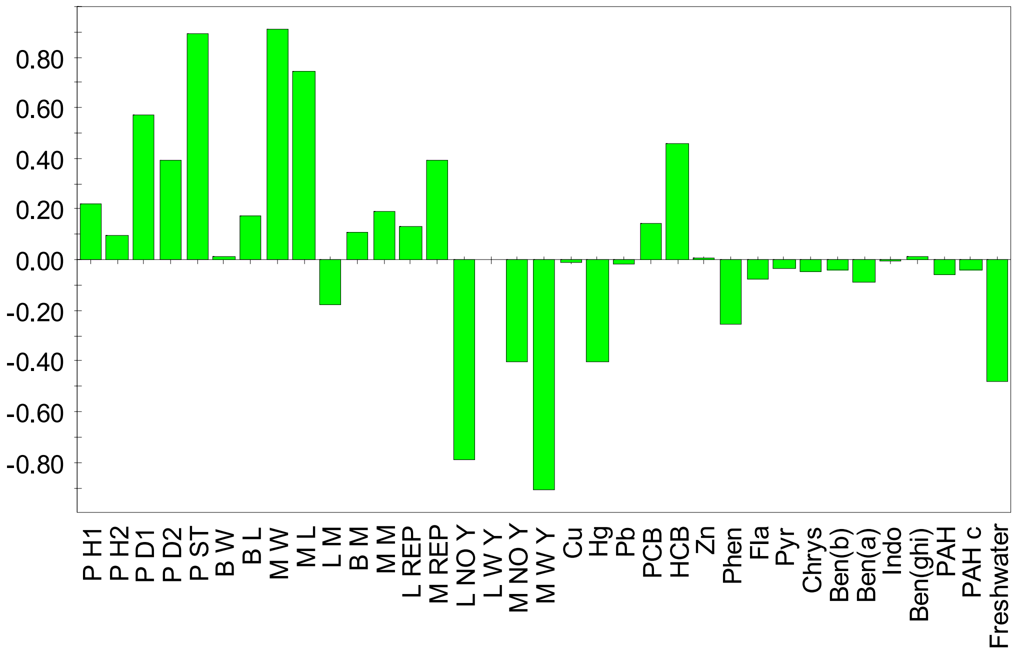

Figure 4. The difference in toxic and reproductive effects and concentrations in sediment, in a comparison between Halls Holme and Krutholmen (positive bars high values at Halls Holme, negative bars high values at Krutholmen).

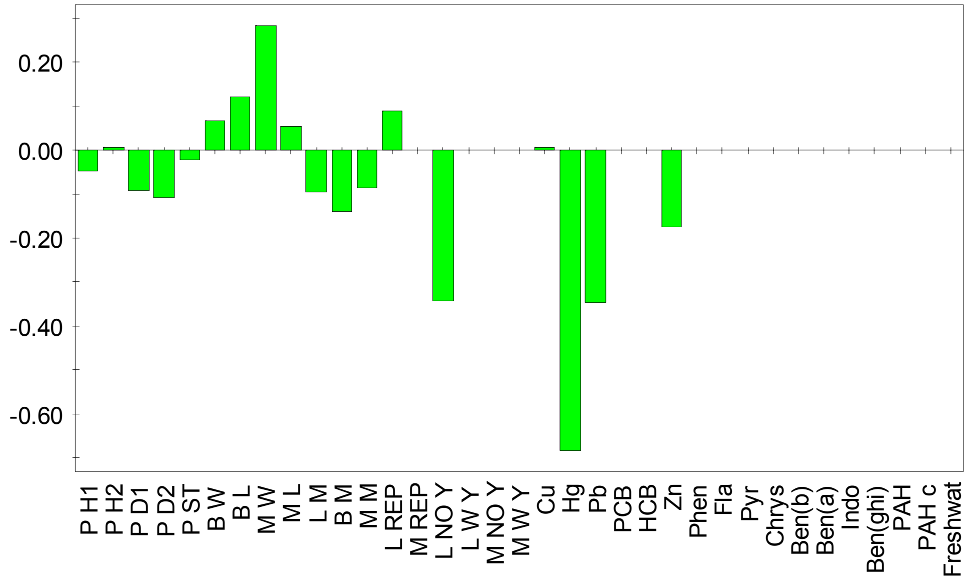

Figure 5. The difference in toxic and reproductive effects and concentrations in sediment, in a comparison between Kiholmen (freshwater reference) and Krutholmen (positive bars high values at Kiholmen, negative bars high values at Krutholmen). This graph shows less bars than the others do because organic pollutants was not analysed.

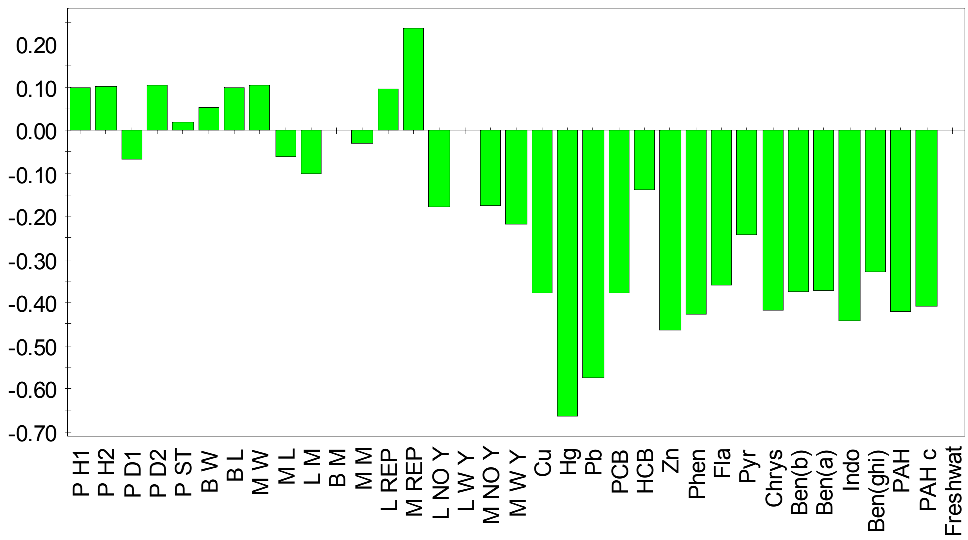

Figure 6. The difference in toxic and reproductive effects and concentrations in sediments in a comparison between Linasundet and Krutholmen (positive bars high values at Linasundet, negative bars high values at Krutholmen).

PCY

A PCY model was based on all the Y-parameters, representing the toxicity and reproduction success. The model is composed of three principal components and explains 77 % of the variance in the Y-parameters. The score- and loading plots for the PCY-model are shown in figures 7 and 8. The score plot shows the locations in relation to each other based on the data of all toxicity variables. The loading plot shows the toxicity and reproductive variables in relation to each other based on data from all locations.

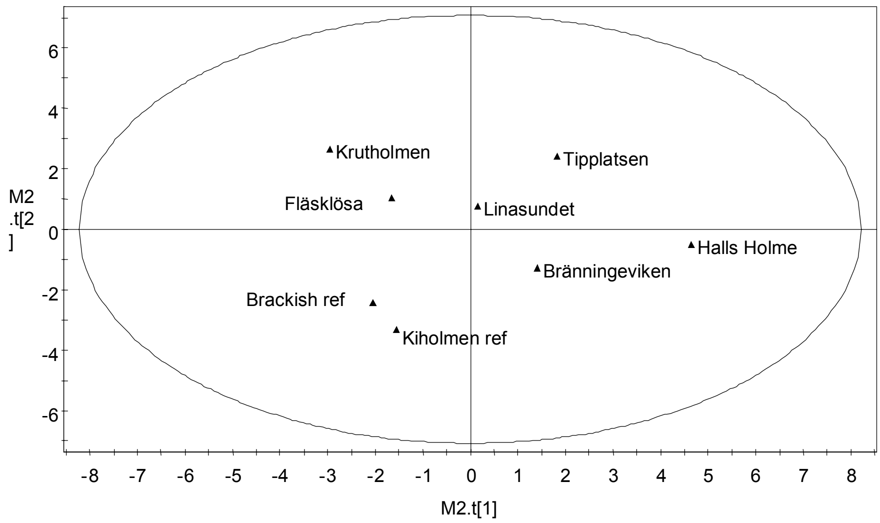

Figure 7. Score plot based on all Y-parameters.

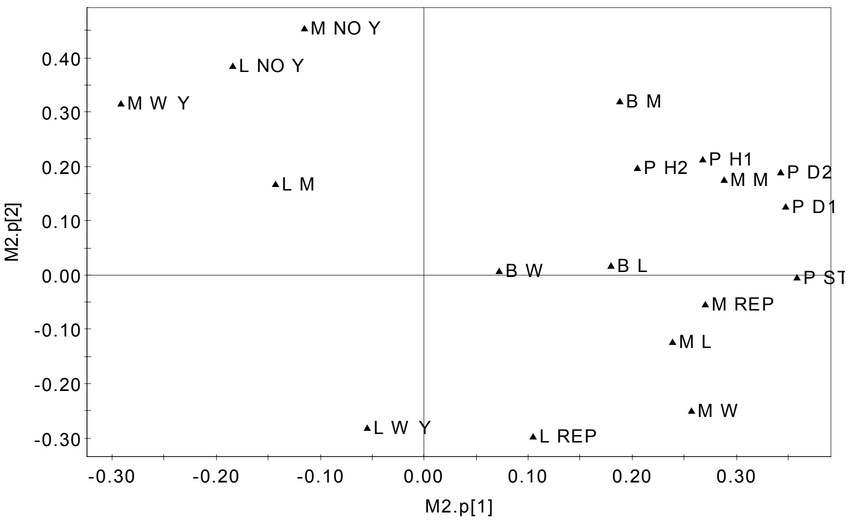

Figure 8. Loading projection showing correlations between Y-parameters.

PLS

First a PLS model was made containing all Y-parameters for screening purposes to get an idea about which Y-parameters whose variation can be modelled based on the pollutants included in the matrix.

Based on the results from the screening and the aim to investigate as many species as possible, four Y-variables (toxicity variables) representing all investigated locations were chosen for the further PLS- evaluation, **P H 2** (***p****erch* embryo **h**atching replicate **2**), **P D 2** (***p****erchlarvae* **d**eformation replicate **2**), **M W** (adult ***mysis*** **w**eight), **L NO Y** (***l****ymnea* **n**umber of **y**oung). PLS 1-models (one Y) were made for each Y-parameter except for the *perch* variables where a PLS 2-model (two Y) were made.

To be able to see more in detail how the X-parameters affect the Y-parameters the coefficients were plotted for each Y-parameter. The Y-parameters can be seen as an equation:

$Y = \beta 1*X1 + \beta 2*X2 + \beta 3*X3 + \ldots\ldots. + \varepsilon$

In the equation the coefficients for the X-parameters are named β. ε is an uncertainty factor.

The *perch* variables (**P H 2** and **P D 2**) are modelled together in a PLS 2-model. Both variables have been transformed (power (–1)) to achieve normal distribution. Due to the transformation the negative coefficients correlate with a toxic effect on Y and vice verse. The results are presented in figures 9 and 10. The model explains 95 % of the variance of the Y-variables and the predicted fraction is 71 %.

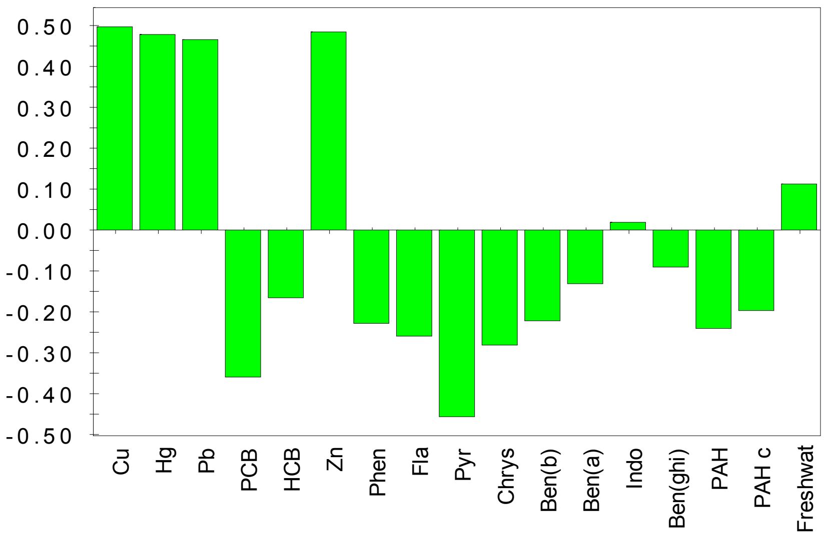

Figure 9. The coefficients explaining the variance in hatching success of Perch embryos, **P H 2** (negative coefficients correspond to a toxic effect).

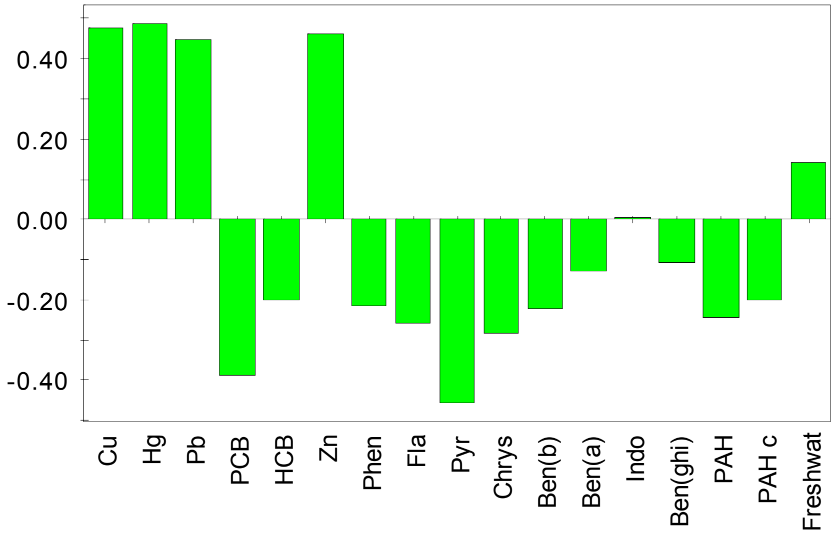

Figure 10. The coefficients explaining the variance in deformation frequens of hatched Perch larvae, **P D 2** (negative coefficients correspond to a toxic effect).

The PLS 1 model for **M W** consists of three components and explains 94 % of the variance of **M W**. The results are shown in figure 11. The predicted fraction of the model is 74 %.

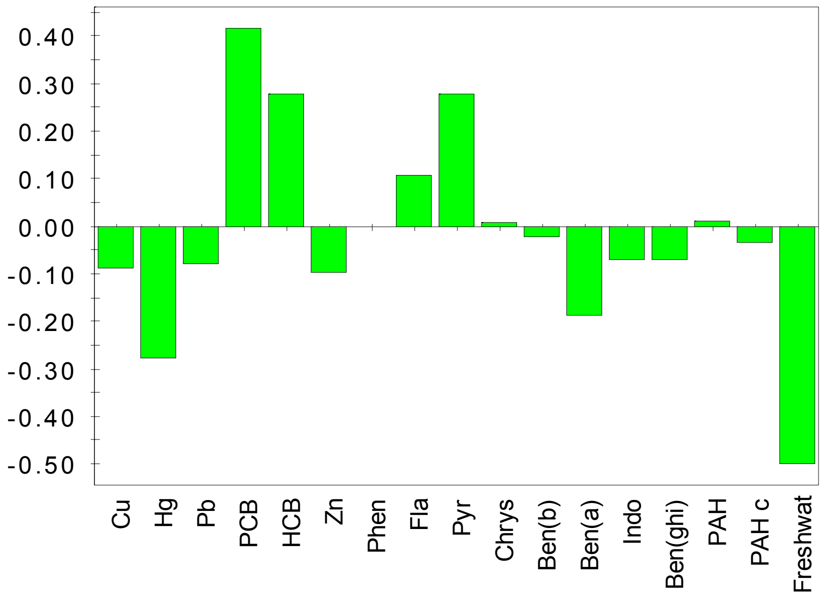

Figure 11. The coefficients explaining the variance in measured weight of adult Mysis, **M W** (positive coefficients correspond to a toxic effect).

L NO Y was first transformed (log) to achieve a normal distribution. The PLS 1 model for **L NO Y** is composed of three components and explain a total of 91 % of the variance in **L NO Y**. The model´s predicted fraction is 61 %. The results are shown in figure 12.

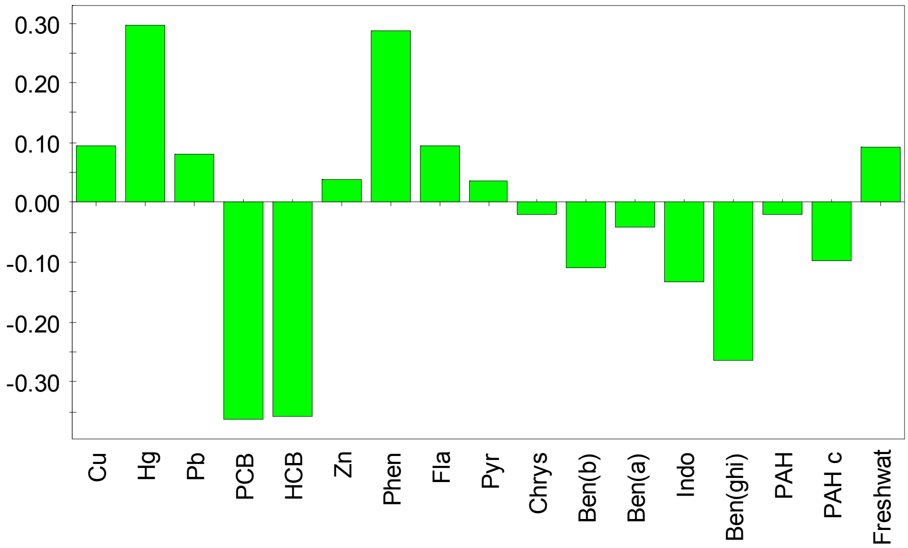

Figure 12. The coefficients explaining the variance in reproduction of Lymnea expressed as number of produced young, **L NO Y** (positive coefficients correspond to a toxic effect).

Discussion

PCA

The results of the PCA model are shown in figures 2 and 3. Figure 3 shows the variable projection based on all included variables. All pollutants vary mainly in the first PC. All pollutants except HCB and PCB are formed in a cluster, showing that they correlate well with each other. This means that at locations where one pollutant have a high concentration the other pollutants are also present in a high concentration and vice versa. There are no response parameters (Y-parameters) in the cluster.

According to the diagram in figure 2, both Halls holme and Krutholmen have the highest toxic effect and/or have the highest contents of pollutants since they are explained to a higher extent, compared to the other locations, by the first and second PC. The Brackish reference location show a low toxicity and a low content of pollutants since it varies in the opposite direction.

The location of Halls holme and Krutholmen in figure 2 compared to figure 3 shows that almost all parameters have high values in these locations. When comparing sediments from Halls holme and Krutholmen (figure 4), the effects from sediments from Halls holme has a higher pronounced acute toxicity than Krutholmen for almost all parameters including adult animals. Sediment from Krutholmen causes stronger sublethal effects registrated as reproductive disorders and survival alternative growth, of young *mysis* and *lymnea*. The concentrations of analysed pollutants are in the same order for the two locations.

According to figure 5, Kiholmen has a lower concentration of mercury and lead than Krutholmen. The toxicity for the parameter L NO Y is also lower at Kiholmen. The toxicity for M W is higher at Kiholmen but except for that there are no great differences between the two locations.

When comparing Krutholmen with Linasundet in figure 6 it is showed that all pollutant concentrations are lower in Linasundet. The toxicity for both young *mysis* and *lymnea* are lower at Linasundet while the toxicity for the rest of the toxicity parameters are slightly higher.

PCY

The results of the PCY model are shown in figure 7 and 8.

The diagram in figure 5 shows the relations between the locations based on all measured effects. Out of all locations Halls holme is the one showing the highest effect on the organisms since it varies in the direction of most toxicity parameters (figure 7). This diagram shows that Krutholmen generally does not have a high toxicity of the species included in this evaluation. It is also indicated that when Halls holme has a high toxicity on certain parameters these parameters have low toxicity at Krutholmen since the two locations vary in opposite directions.

The references are placed closed together in the diagram in figure 7. They both have a low toxicity together with Krutholmen.

According to the diagram in figure 8 most of the *perch* variables are correlated to each other. The variables for the other species do not show any clear correlations within the species. Four of the *mysis* variables (concerning adults) are placed on one side of the diagram while the remaining two (concerning young *mysis*) are placed on the opposite side. This means that the *mysis* toxicity variables for adults and young *mysis* are anti-correlated to each other. When the toxicity for adults are high the toxicity for young *mysis* are low and vice versa. There may be different theories to explain this. The *bufo* variables are placed on one side of the diagram. There may be some kind of correlation between the *bufo* variables but it is not very clear. The *lymnea* variables show no clear correlations within the species.

PLS

The number of locations included in this evaluation is limited why the results of the evaluation should be considered to be the best possible for the aera.

The *perch* variable can be considered as a representative for the other *perch* variables since all the *perch* variables correlate according to the PCY-model. The *mysis* variable can in the same way be considered to represent the other adult *mysis* variables. The young *mysis* variables are affected the opposite way. The *lymnea* variable doesn't represent the *mysis* species in general.

Tries were made to make PLS models for the *bufo* variables. However it didn't result in any successful models and the *bufo*-parameters remain unmodelled.

The coefficient plots show which pollutant variables that are important to explain the variance in the Y-parameters. According to figure 9 the toxicity variable *perch* hatching is affected by PAH but not by metals. The other *perch* variable, *perch* deformation frequency, is effected in the corresponding way (figure 10).

According to figure 11, *mysis* weight is affected mainly by PCB's, HCB, Flouranthene and Pyrene. The metals do not seem to have any toxic effect together with the remaining PAH. The toxicity for *mysis* weight is lower in freshwater and higher in brackish water. The opposite conclusions are valid for the young *mysis* variables.

The metals and Phenanthrene adversely affect the number of young lymnea. The PCB's and HCB together with most of the remaining PAH have no toxic effect. This *lymnea* variable is not representing the effects on any of the other *lymnea* variables.

Conclusions

The number of observations included in the models are relatively few. This has to be considered when studying the results from this evaluation. If a new location (observation) was added to the matrix the results could turn out different. The low number of observations makes the matrix sensitive.

The PLS models are all fairly good both in explanation and predicted fraction. When the number of observations is limited and the number of parameters are large, correlations may be found between X- and Y-parameters by a coincidence. This also has to be considered when studying the results. If a comparison of the results for the different kind of organisms it can be summarized as follows.

One group of animal, the shrimps, represented by adult *mysis*, were differently affected in fresh water and brackish water. The toxicity is more pronounced in brackish water.

PCB's, HCB and Pyrene affect the adult mysis mainly. The opposite is valid for young *mysis*

The *perch* variables, representing fish parameters, are mainly affected by the PAH.

The *lymnea* variables, representing the group of molluscs, are not correlated well to each other why the results from the evaluation of the number of young *lymnea* is not considered to be valid for all *lymnea* variables. Metals and Phenanthrene and Flouranthrene affect the number of young lymnea.

It was not possible to explain the variation of the *bufo*-variables based on the pollutants included in the model.

Non of the effects registrated on organisms included in the multispieces testsystem could be related to the mercury content in the sediments.

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IVL Svenska Miljöinstitutet AB
IVL Swedish Environmental Research Institute Ltd

P.O. Box 210 60, SE-100 31 Stockholm
Visit: Hälsingegatan 43, Stockholm
Phone: +46 8 598 563 00
Fax: +46 8 598 563 90

P.O. Box 5302, SE-400 14 Göteborg
Visit: Aschebergsgatan 44
Phone: +46 31 725 62 00
Fax: +46 31 725 62 90

Aneboda, SE-360 30 Lammhult
Aneboda, Lammhult
Phone: +46 0472 26 77 80
Fax: +46 472 26 77 90

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