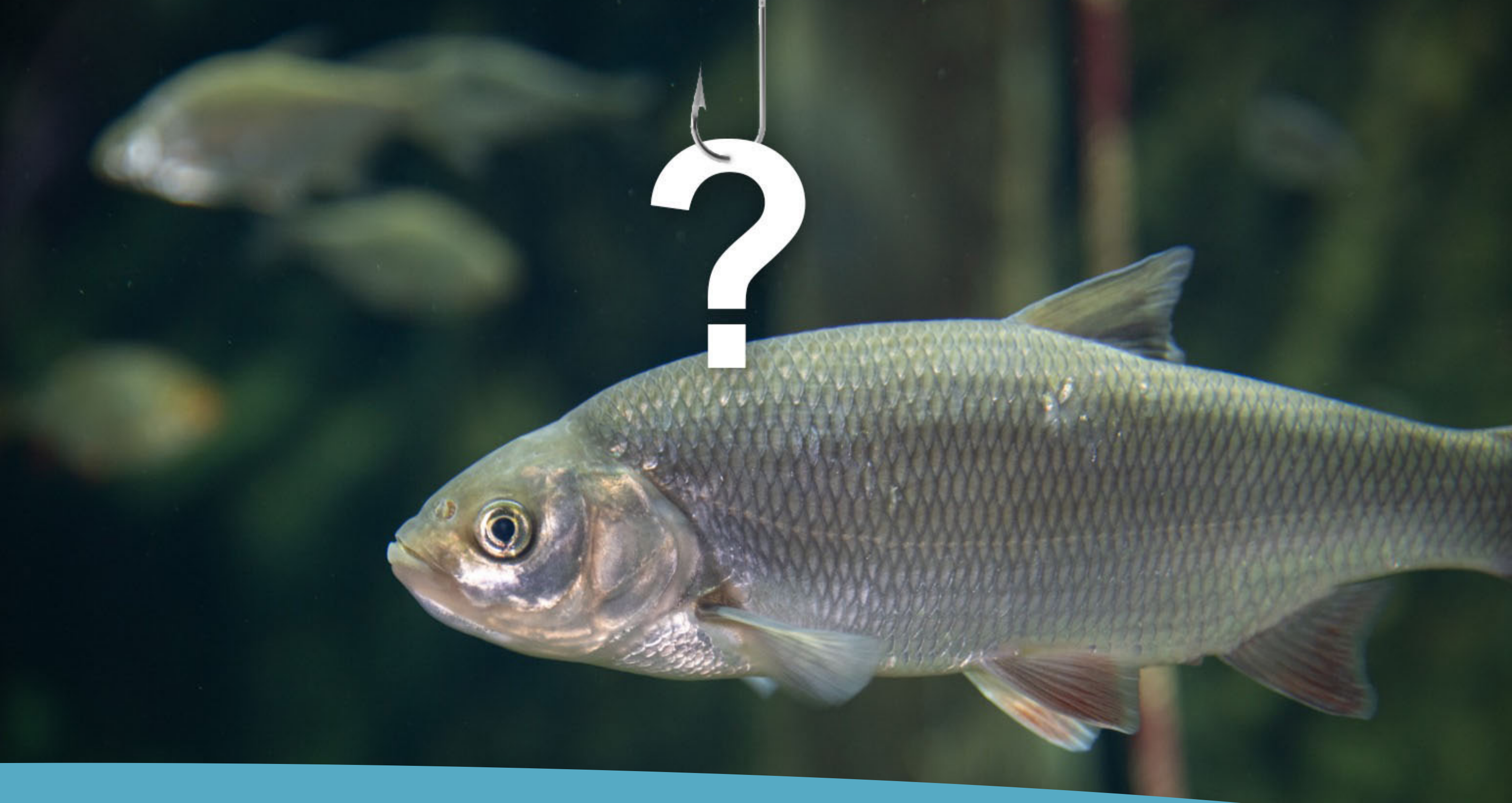

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Fishing cyprinids for food

Evaluation of ecosystem effects and contaminants in cyprinid fish

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Sammanfattning

Vår slutsats är att ett riktat karpfiske för matfisk vid kusten kan ha positiva effekter på ekosystemet och fångsten är utifrån miljögiftshalter lämplig för konsumtion. Det finns dock ett vetenskapligt behov av utvärdering och kvantifiering av ekosystemeffekter vid en uppskalning av riktat karpfiske. En uppskalning kräver också en ökad hantering och försäljning av karpfisk vilket i sin tur kräver förändringar i konsumenternas attityd till karpfisk och nya livsmedelsprodukter av karpfisk.

Flera tidigare kommersiellt viktiga vilda fiskbestånd i Östersjön är i dåligt skick med låga populationsstorlekar och småvuxna individer. Dagens svenska livsmedelskedja är därför starkt beroende av odlad och importerad fisk. Samtidigt har eutrofiering och klimatförändringar lett till ökande bestånd av karpfiskar (fiskar i familjen cyprinder, t.ex. braxen, mört, id) i många kustområden i Östersjön, vilket hotar att nationella och internationella miljömål inte uppfylls. Under de senaste åren har intresset för att fiska karpfisk som livsmedel ökat i Finland och Sverige. Denna rapport utvärderar potentiella ekosystemeffekter från ökat riktat karpfiske för mat, och hur vi kan övervaka och bedöma dessa effekter i Östersjön. Vi bedömer också potentiella hinder för ökat karpfiske för konsumtion i form av livsmedelssäkerhet på grund av miljöföroreningar och marknadsincitament för fiskare.

En litteraturöversikt av reduktionsfisken av karpfisk i sjöar visar att åtgärden har haft önskvärda effekter på vattenkvalité i cirka 60% av de fall där den har testats. I Östersjön har dock endast ett fåtal pilotprojekt för reduktionsfiske på karpfisk genomförts. Skillnader mellan kustområden och sjöar gör det osannolikt att samma grad av framgång som i sjöar också skulle gälla kustområden, särskilt om man mäter effekten främst som lägre halter av näringsämnen i vattnet. Ändå tror vi att ett hållbart fiske riktat mot karpfisk skulle kunna ge vissa positiva effekter på vattenkvalitet och makrofyter i Östersjön i ett längre tidsperspektiv. I linje med vår litteraturöversikt föreslår vi ett övervakningsprogram för utvärdering av ett riktat karpfiske i kustområden. Baserat på de potentiella ekosystemeffekterna föreslår vi att övervakning bör prioritera fisksamhällets sammansättning, siktdjup och klorofyll α, och makrofyter. För mer noggranna vetenskapliga utvärderingar bör även övervakningsprogram innehålla analyser av växt- och djurplankton, samt och närings- och syrehalter.

Ett ökat fiske på karpfisk från Östersjön som livsmedel kan också få samhällseffekter. För att undersöka om mänsklig konsumtion av karpfisk medför någon ökad risk för exponering för miljögifter analyserade vi koncentrationen av kvicksilver, kadmium, dioxiner, PCB, PFAS och PBDE i braxen, id och mört från fem platser längs den svenska kusten i Östersjön. Våra resultat visar att det, baserat på de nuvarande regler och rekommendationer i Sverige, inte finns några uppenbara hälsorisker med att äta karpfisk fisk från Östersjön åtminstone någon gång i veckan. Eftersom kunskap och regler om vissa miljögifter är dåliga eller obefintliga, anser vi dock att det är viktigt att göra en mer omfattande studie, särskilt för PFAS. Ett ökat fiske riktat mot karpfisk skulle också diversifiera det småskaliga kustfisket i Sverige, men den låga efterfrågan gör det riskabelt för fiskare att investera i redskap och distribution blir relativt dyr.

Abstract

We conclude that a coastal cyprinid fishery may have positive effects on the ecosystem and with regard to levels of toxic contaminants, the fish is safe for humans to eat. There is, however, a need to scale up the targeted cyprinid fishery in order to evaluate and quantify the effects on the ecosystem. Scaling up from the pilot scale fisheries requires a change in consumer's attitude and product development, so that larger quantities of cyprinid fish can be harvested and sold.

Several wild fish stocks in the Baltic Sea are in poor condition and today the supply of fish for human consumption in Sweden heavily relies on farmed and imported fish. At the same time, eutrophication and climate change has led to increasing populations of cyprinid fish (e.g. bream, roach, ide) in many coastal areas of the Baltic Sea, which threatens to violate Swedish and international environmental goals. During recent years, there has been an increased interest to fish cyprinids for human consumption in Finland and Sweden. This report evaluates potential ecosystem effects resulting from an increased cyprinid fishery, and how to monitor and assess these effects in the Baltic Sea. We also assess potential barriers to increased cyprinid fisheries for human consumption due to food safety issues resulting from environmental contaminants and market incentives for fishers.

In a literature review on biomanipulation targeting cyprinids in lakes, we show that removing cyprinids as a measure to improve water quality has been successful in around 60% of the cases where it has been tested. In the Baltic Sea, however, there have only been a few pilot projects of biomanipulation of cyprinids. Differences between coastal areas and lakes makes it unlikely that the same success rate as in lakes would also apply to coastal areas, especially when considering lowering of nutrient concentrations. Still, we think that a sustainable fishery targeting cyprinids may promote at least positive effects on water transparency and macrophytes in the Baltic Sea on a longer time-scale. In line with results from our literature review, we suggest a monitoring program for evaluation of a targeted cyprinid fishery in coastal areas. Based on the potential ecosystem effects of a cyprinid fishery we suggest that monitoring should prioritize fish community composition, water transparency, chlorofyll α, and submerged macrophytes. For more thorough scientific evaluations, the monitoring program should also include abundance of phyto- and zooplankton, as well as and nutrient and oxygen levels.

An increased use of cyprinids from the Baltic Sea as human food will also have societal impacts. To examine if human consumption of cyprinid fish entails any increased risk of exposure to contaminants, we analysed concentration levels of several toxins (mercury, cadmium, dioxins, PCBs, PFAS and PBDE) in bream, ide and roach from five sites along the Swedish coast of the northern Baltic Sea. Our results show that, based on the regulations in Sweden today, cyprinids meet all health regulations for human food. Based on recommendations of weekly intake there are no apparent health risks of consuming cyprinid fish from the Baltic Sea at least weekly. However, since knowledge and regulations of certain environmental toxins are poor or non-existent, we believe it is important to conduct a more comprehensive study, especially for PFAS. An increased fishery targeting cyprinids would diversify the small-scale coastal fishery in Sweden, but the currently low demand makes it risky for fishers to invest in equipment and distribution becomes relatively expensive.

Preface

This report is summarizing a project financed by the Swedish Agency for Marine and Water Management with the aim to investigate the environmental and societal effects of a targeted cyprinid fishery in coastal areas of the Baltic Sea for food production. The project complements a recent SLU project, where potential indicators to assess the status of currently low exploited fish were reviewed and suggested (Sundblad et al. 2020). During progress of this report, we have collaborated with the NGO 'Race For The Baltic' for sampling of fish and market perspectives of cyprinid fish for human consumption.

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1. Introduction

Several important and commercially exploited fish stocks in the Baltic Sea are overexploited (HELCOM 2018a; ICES 2019, 2021). As a consequence, there is a low profitability in coastal fisheries and the number of small-scale coastal fishers have decreased (Andersson 2019; Waldo & Lovén 2019). Instead, farmed and imported fish for human consumption has increased last decades (Ziegler & Bergman 2017).

Despite the HELCOM Baltic Sea Action Plan (BSAP) and associated efforts to reduce nutrient loads, eutrophication is still a major environmental problem in the Baltic Sea (HELCOM 2018a). Cyprinid fish species like roach (*Rutilus rutilus*), bream (*Abramis brama*), and silver bream (*Abramis bjoerkna*) thrive in eutrophic waters and their abundances are increasing in many parts of the Baltic Sea (HELCOM 2018a). High abundances of cyprinid fish may even accelerate eutrophication-associated symptoms, such as algal blooms and oxygen depletion, as cyprinids prey on zooplankton and zoobenthos, and recycle nutrients back to the water column through resuspension of phosphorus particles from the sediment (Adamek & Marsálek 2013). Despite high abundance of cyprinids in the Baltic Sea, there is today a limited fishery on cyprinids for human food consumption in Sweden. Instead, there has been local short-term attempts to reduce populations of cyprinid fish in some coastal bays to reduce eutrophication and improve ecological status. However, in recent years there have been increased efforts to promote cyprinid fisheries (mainly bream and ide (*Leuciscus idus*)) in Sweden, aiming at developing a future food resource in Sweden with potential positive effects on the environment.

Pauly et al. (1998) warned that fisheries are “fishing down the food web”. Fisheries aim for the fish species at the top of the food web but once they are reduced, fisheries target the next lower trophic level and so on, preventing recovery of higher trophic levels, until degraded aquatic ecosystems only consist of small-bodied lower trophic-level fish and invertebrates. Without management targets and a long-term sustainable use, an increased focus of fishing on cyprinids might hence be just another step down the food-web, preventing recoveries of piscivore fish. Understanding and monitoring the ecosystem effects of an increased cyprinid fishery as well as the impact from the fishery on targeted stocks and populations is therefore essential.

One challenge using Baltic Sea fish for human consumption is the potential health risks posed by hazardous substances (Hallikainen et al. 2011). Thus, before fisheries increasingly targeting cyprinids are developed, there is a need to determine and monitor if the concentration of hazardous substances in the fish are below the target levels set for human consumption.

This report reviews current knowledge and identifies knowledge gaps to understand the opportunities and limitations of an increased targeted cyprinid fishery in the Baltic Sea. A cyprinid fishery could be a means to increase Sweden´s degree of food self-sufficiency using local and previously non-targeted fish stocks. The report includes an overview of ecosystem effects resulting from biomanipulation of cyprinid fish, mainly from freshwater lakes but also from the Baltic Sea. We also analyse concentrations of contaminants in different cyprinid species from the Baltic Sea. The overarching aim of the report is to suggest a monitoring program for evaluation of current and future fisheries targeting cyprinid fish. Finally, we identify factors that could limit the development of a cyprinid fishery for human consumption.

2. Cyprinids as food and in the environment

The cyprinids (Cyprinidae), also called carp fish, is the most species-rich family of freshwater fish in northern Europe, and are important food sources for humans across Eurasia. To strengthen the food supply chain, there is a rising interest from managers, authorities and fishers to utilise more cyprinid fish in Sweden. In this report, we focus on bream, roach and ide as potentially species for a future targeted fishery on cyprinids. These species are covered relatively well in the current resource and environmental monitoring in Nordic waters (Sundblad et al. 2020), and with few exceptions categorized as least concern species on the Swedish IUCN Red List (SLU Artdatabanken 2020).

2.1. Cyprinids as a food resource

Cyprinids can be considered a relatively ‘climate smart’ food resource. The carbon dioxide footprint from cyprinid fisheries in Swedish lakes has an average carbon footprint of 0.77kg CO_2 equivalents per kg of edible (cyprinid) product (Hornborg & Främberg 2020). As the fisheries in the Baltic Sea will use similar (static) gear, a similar carbon footprint for marine cyprinids can be expected. This is about one third of the greenhouse gas emissions of fisheries targeting North Sea Atlantic cod and farmed (Norwegian) Atlantic salmon, and 97% less compared to the production of beef in Sweden (RISE Climate Database 2018).

Increased utilization of cyprinids as seafood in Sweden may also improve nutritional security, as freshwater bream has higher concentration of vitamin B12, selenium and niacin compared to Atlantic cod (*Gadus morhua*) and farmed Atlantic salmon (*Salmo salar*; Hornborg et al. 2019). In addition, the concentration of polyunsaturated fatty acids is substantially higher in for example in bream compared to Atlantic cod.

2.2. Eutrophication and climate change benefit cyprinids

Eutrophication in the Baltic Sea is caused by historic external loading of nitrogen and phosphorus from agriculture, sewage, industry and aerial deposition (Smith 2003; Diaz & Rosenberg 2008). Nutrient loads have decreased during recent decades, but water column nutrient concentrations remain elevated (HELCOM 2018a). This eutrophic state has resulted in many areas, especially in the southern and east part of the Baltic Sea, in high phytoplankton biomass with summer blooms of blue-green algae and oxygen deficiency (Horne & Goldman 1994; Iho et al. 2017). Eutrophication has changed the structure and function of Baltic Sea coastal fish communities, and a typical response to eutrophication is an increased abundance, both in numbers and biomass, of cyprinid fish (Neuman 1987; Sandström & Karås 2002; Olin et al. 2002; Bergström et al. 2016, 2018; HELCOM 2018a).

Cyprinids are highly capable of consuming plant material and zooplankton in turbid and eutrophic waters (Lammens et al. 1987; Vinni et al. 2000), and to reproduce in a range of different habitats (Barthelmes 1983). Furthermore, the predation pressure on cyprinids might be reduced in turbid and eutrophic waters, since the foraging capacity of piscivores is reduced in such conditions (Persson et al. 1991). The changing climate in the Baltic Sea region, with shorter and lowered distributions of ice cover and long-term increasing trends in water temperature is predicted to further favour cyprinid fish (Olsson et al. 2012; Bergström et al. 2016, 2018; Östman et al. 2017; HELCOM 2021).

2.3. Feeding behaviour and resuspension

Benthic fish, such as bream, disturb the sediment surface in their search for food, which causes resuspension of sediment particles into the water column that increases nutrients availability and the productivity of the system, increasing turbidity (Breukelaar et al. 1994; Vanni 2002; Adamek & Marsálek 2013). Bioturbation of the sediment also increases the risk of wind-, wave-, and ship-induced turbidity, and makes it more difficult for submerged vegetation to establish. High turbidity reduces the water clarity, which in turn makes foraging for visual hunters such as pike (*Esox lucius*) and perch (*Perca fluviatilis*) more difficult, whereas cyprinids are able to forage effectively in low light conditions (Diehl 1988). Using a model, Meijer et al. (1990) suggested that more than 50% of the turbidity of eutrophic lakes could be attributed to sediment resuspension, mainly caused by benthivorous fish. Fish can also play a significant role in nutrient dynamics by making phosphorus and nitrogen bio-available through excretion, enhancing ecosystem productivity and algal growth (Vanni 2002).

2.4. Nutrient reduction by targeted cyprinid fishing

Fishing can indirectly contribute to phytoplankton growth through a trophic cascade if fishing depletes the large predatory fish, which subsequently reduces the predation pressure on zooplanktivorous and benthivorous fish like cyprinids. Increased biomass of cyprinids can reduce zooplankton abundance, which in turn increases the abundance of phytoplankton that serve as food for zooplankton (Carpenter et al. 1985; Pauly et al. 1998; Österblom et al. 2007; Eriksson et al. 2009). Thus, a negative relationship between cyprinids and water quality is often observed (Horppila & Kairesalo 1990; Hansson et al. 1998). A targeted cyprinid fishery could reduce eutrophication symptoms by decreasing the rate of resuspension of nutrients from the sediment, and reducing the phytoplankton biomass via increased grazing by zooplankton (Bernes et al. 2015).

Fishing with the short-term aim to rapidly reduce benthivorous and planktivorous fish to lower the predation pressure on zooplankton, is called 'reduction fishing' or 'biomanipulation' (Shapiro et al. 1982). Biomanipulation in lakes is typically conducted by mass removal of cyprinid fish (Olin et al. 2006), but has also been performed in concert with stocking of piscivorous fish (Shapiro & Wright 1984). These food-web manipulation methods have been carried out since the 1980s as a method of ecological restoration in eutrophic lakes and ponds (Appendix 1). A large number of biomanipulation studies have been conducted in eutrophic lakes and ponds (Table A1.1-A1.6 in Appendix 1), providing extensive experiences on the ecosystem effects of removing cyprinid fish biomass.

Cyprinid fish contain 0.7 to 0.8% of wet weight (ww) phosphorus (P) depending on species, and 2.5% nitrogen (Mäkinen 2008). A catch of 100 kg ha^{-1} and an average P content of 0.75% of the fish would thus remove the equivalent of 0.75 kg phosphorous per hectare in a typical lake. If fishing is conducted in the spring targeting spawning fish, the P-content could be higher (Kitchell et al. 1975) as eggs and sperm may be richer in phosphorus.

2.5. The potential of a targeted cyprinids fishery on ecosystem restoration

Human activities such as extensive fishing, high loads of nutrients and contaminants loads, and climate-related impacts, might decrease the provisioning services of the Baltic Sea ecosystem (Bryhn et al. 2015; Blenckner et al. 2021). Fisheries targeting cyprinids may not only reduce internal nutrient concentrations through decreased recycling of nutrients, but might also contribute to ecosystem restoration or maintenance of the several ecosystem services:

- Improving water quality and transparency

Effective biomanipulation of cyprinids affects the food-web from fish to algae and water quality. Mass removal of cyprinids can lead to decreased sediment resuspension and reduced grazing pressure on zooplankton, which in turn lowers the abundance of phytoplankton and particles in the water, hence, increasing water transparency and recreational values (Mehner et al. 2002). On a longer time-scale the fish community might shift back to a dominance of predatory fish and species favoured by less eutrophic conditions (Søndergaard et al. 1998; Jeppesen et al. 2002; Liboriussen et al. 2007).

- Improving habitat quality

By decreasing the turbidity, light conditions improve and increase the establishment and distribution of submerged macrophytes, which in turn stabilize the sediment and contribute to nutrient sedimentation (Duarte 2000). An increase of macrophytes has also positive effects on water transparency and provides habitat for fish and birds (Hanson & Butler 1994). Predatory fish benefit from submerged vegetation and can in turn stabilize clear water conditions through predation on cyprinids (Carpenter et al. 1985; Diehl 1988).

- Human food supply

Cyprinids from the Baltic Sea are potentially suitable as human food. As such, they represent an alternative source of protein, and if exploited will benefit small scale coastal fisheries. Bonow and Svanberg (2013) stress that there has been a tradition of consuming cyprinids (mainly bream and ide) in Sweden during the end of the 1800s.

2.6. Cyprinids and environmental goals

As cyprinid fish both respond to eutrophication and climate change and may have an important impact on ecosystem functions (Olin et al. 2002; Bergström et al. 2016, 2018; HELCOM 2018a; Blenckner et al. 2021), cyprinid abundance is an indicator of ecosystem status within the Marine Strategy Framework Directive (MSFD; HELCOM 2018a). Because cyprinids can have detrimental impact on ecosystem quality but are important as prey for piscivorous fish, the abundance indicator of cyprinid fish should be within a range to be considered as a good environmental status. In an assessment until 2016, cyprinid abundance met the requirements for good environmental status in the majority of areas surveyed, but not in all due to too high abundances (HELCOM 2018a, Fig. 1). Hence, Sweden largely fulfilled the goals within the MSFD indicating no or only local actions towards cyprinids were required.

Figure 1: Status assessment of the cyprinid/mesopredator indicator in Baltic Sea coastal fish until 2016. Green dots indicate monitoring sites where abundance was considered as "Good environmental status", and red dots/areas indicating "Not good ecological status". Figure from HELCOM 2018a.

The next assessment of cyprinid abundance will be after 2022, but an intermediate assessment using data until 2019 indicates that cyprinid abundance is increasing in some areas in the Swedish part of the Baltic Sea (Table 1). Roach is the single most common species in this indicator, and bream and ide only constitute a smaller part. It is not known to what degree the population dynamics between different cyprinid species are synchronous over time, but a targeted cyprinid fishery could contribute to locally achieving environmental goals according to the MSFD.

Table 1: Output from HELCOM's intermediate assessment (HELCOM work in progress) of coastal fish indicators for the functional group 'Mesopredators' (i.e. cyprinids) at different fish monitoring sites in the Swedish part of the Baltic Sea. Change indicate the change in the indicator 'Abundance of cyprinids' since the last assessment 2016.

Sub-basin	**Monitoring area**	**Change**
Bothnian Bay	Råneå	No change
Bothnian Bay	Kinnbäcksfjärden	No change
The Quark	Holmön	Increase
The Quark	Norrbyn	Increase
Bothnian Sea	Gaviksfjärden	Increase
Bothnian Sea	Långvindsfjärden	No change
Bothnian Sea	Forsmark	Decrease
Åland Sea	Lagnö	Increase
Northern Baltic Prope	Askö	No change
Western Gotland Basin	Kvädöfjärden	No change
Western Gotland Basin	Vinö	Increase
Bornholm Basin	Torhamn	Decrease

According to national management plans, fish stocks should be harvested at sustainable levels and have a "natural-like" size distribution (Östman et al. 2016). Cyprinids are together with perch and the European- and Baltic flounder (*Platichthys* sp.) included as indicator of status assessments for coastal fish within the MSFD (HELCOM 2018a) such that their abundance and population dynamics is also related to environmental goals and national management plans. As the cyprinid abundance should be within a range for achieving good status, very high cyprinid catches in a region could jeopardise the MSFD goals. In general, we do not see any major conflict between these goals for cyprinids, but rather the opposite, as a sustainable harvest of cyprinids may contribute to cyprinid abundance indicators reaching the range significant for achieving good ecological status.

3. Knowledge of ecosystem effects of a targeted fishery on cyprinids

3.1. Current knowledge of biomanipulation targeting cyprinids in freshwaters

There are several reviews published on biomanipulation of cyprinids in freshwater ecosystems (e.g. Benndorf 1990; Jeppesen et al. 1990; Hansson et al. 1998; Meijer et al. 1999; Gulati & Van Donk 2002; Søndergaard et al. 2007; Bernes et al. 2015; Triest et al. 2016), but only few studies from marine environments. We compiled data from 118 studies of biomanipulation of cyprinids in eutrophic lakes in northern Europe (Appendix A1.1-5), of which most were conducted during the 1990´s. The biomanipulation projects were usually carried out during three to four years, where on average 180 kg cyprinids per hectare and year were removed, resulting in cyprinid stock reductions by up to 96%. The study objects include small ponds of a few hectares to lakes and lake-systems several thousand hectares in size, most of them shallow.

The studies can be divided into five categories based on the type of biomanipulation performed:

a) Total fish reductions (elimination of the total fish stock, e.g. by using piscicides),
b) Cyprinid reductions (partial removal of fish stocks - usually cyprinids),
c) Cyprinid reductions and stocking of predatory fish,
d) Cyprinid reductions and chemical (Al, Ca or Fe) or physical treatments (aeration of the bottom layer, dredging or dilution with nutrient-poor water),
e) Stocking of piscivores, without any additional treatments.

Aggregated information about nutrients and chlorophyll α levels before and after the biomanipulations, and the amount of fish removed are presented in Table 2. A summary of the effect sizes estimator and log response ratios for nutrients and chlorophyll α levels are available in Figure 2. Log response ratios (LRR) show how much a variable has changed after a treatment relative the conditions prior to the experiment, calculated as $\log(X_{after}/X_{before})$, where X is the variable of interest, e.g.

nutrient and chlorophyll concentrations. For several parameters (Secchi depth, macrophytes, and zooplankton) it was, however, difficult to calculate LRR as data have been provided in different units or are bounded (Secchi depth by water depth, and macrophytes cover for example between 0 and 100% coverage). Therefore, we instead classified outcomes into discrete classes: Increase, Decrease and No changes. For a variable to change we considered 10% as the error margin and limit of change.

Table 2: Summary of the amount of fish compiled lake restorations in Appendix 1. The studies are divided into methods used for biomanipulation. The amount of removed fish is shown in total per study and total per study and year ± standard deviation (SD), for n studies with available values. The outcome of the measured variables is shown (mean ± standard deviation) before and after the manipulation, and. n = number of studies with measurements both before and after. NA = not applied.

Variable	Outcome	Non-targeted fish reduction	Cyprinid reduction	Cyprinid reduction and piscivore stocking	Cyprinid reduction and chem./phys. treatments	Stocking of piscivores	Total (average)
Fish removal (kg ha^{-1})	total ± SD	416 ± 282	284 ± 222	532 ± 320	559 ± 708	NA	**448 ± 375**
	Years ± SD	1.6 ± 1.2	3.6 ± 2.2	5.2± 4.1	3.2 ± 0.9	NA	3.8 ± 2.9
	total annual ± SD	305 ± 251	97 ± 97	139 ± 118	196 ± 248	NA	**184 ± 179**
	n	**15**	**44**	**33**	**14**	**9**	
Total phos-phorus in water (mg l^{-1})	mean before ± SD	0.24 ± 0.34	0.11 ± 0.11	0.20 ± 0.19	0.14 ± 0.09	0.17 ± 0.16	**0.17 ± 0.18**
	mean after ± SD	0.17 ± 0.23	0.14 ± 0.33	0.14 ± 0.09	0.07 ± 0.05	0.17 ± 0.16	**0.14 ± 0.17**
	n	**11**	**37**	**32**	**12**	**6**	
Total nitrogen in water (mg l^{-1})	mean before ± SD	1.46 ± 1.11	1.55 ± 1.09	2.18 ± 0.70	1.44 ± 0.74	3.65 ± 2.33	**2.06 ± 1.19**
	mean after ± SD	1.24 ± 1.44	1.29 ± 0.71	1.55 ± 0.51	1.12 ± 0.25	1.85 ± 0.50	**1.41 ± 0.68**
	n	**6**	**26**	**21**	**5**	**2**	
Chloro-phyll α (µg l^{-1})	mean before ± SD	86 ± 74	39 ± 174	86 ± 50	54 ± 26	176 ± 128	**88 ± 90**
	mean after ± SD	23 ± 16	41 ± 39	57 ± 40	28 ± 23	81 ± 62	**46 ± 36**
	n	**9**	**27**	**31**	**7**	**5**	

There are no objective criteria to determine if a biomanipulation is successful or not (Hansson et al. 1998). At least two of the variables, concentrations of phosphorus and chlorophyll α and Secchi depth had to improve through the biomanipulation for us to consider it as a successful biomanipulation. If only one variable improved, we considered it as partly successful, and if none of these variables improved or the improvement lasted less than one year, we considered biomanipulation as unsuccessful (Fig. 3).

There is a risk for publication bias due to biomanipulation attempts that have not improved the conditions are not reported, whereas treatments that are more successful have been reported. It is difficult to assess potential bias and our results should therefore been considered as an upper probability or effect of biomanipulations.

Our review shows that biomanipulations of cyprinids are far from always successful (Table 2, Figs. 2-3). Regarding phosphorus concentration, the biomanipulations targeting cyprinids had an average LRR = -0.1, which was not significant from no change (t-test: t = -0.9, df = 35; p = 0.3, Fig. 2). One potential reason for lack of decreasing phosphorous levels may be if moving fishing gear, like seines or trawls, have been used for cyprinid fisheries that disturb the sediment and increase nutrient concentrations in the water column. Results of cyprinid biomanipulations were similar with respect to nitrogen concentrations (LRR = -0.04, t_{24} = -0.63, p = 0.3, Fig. 2).

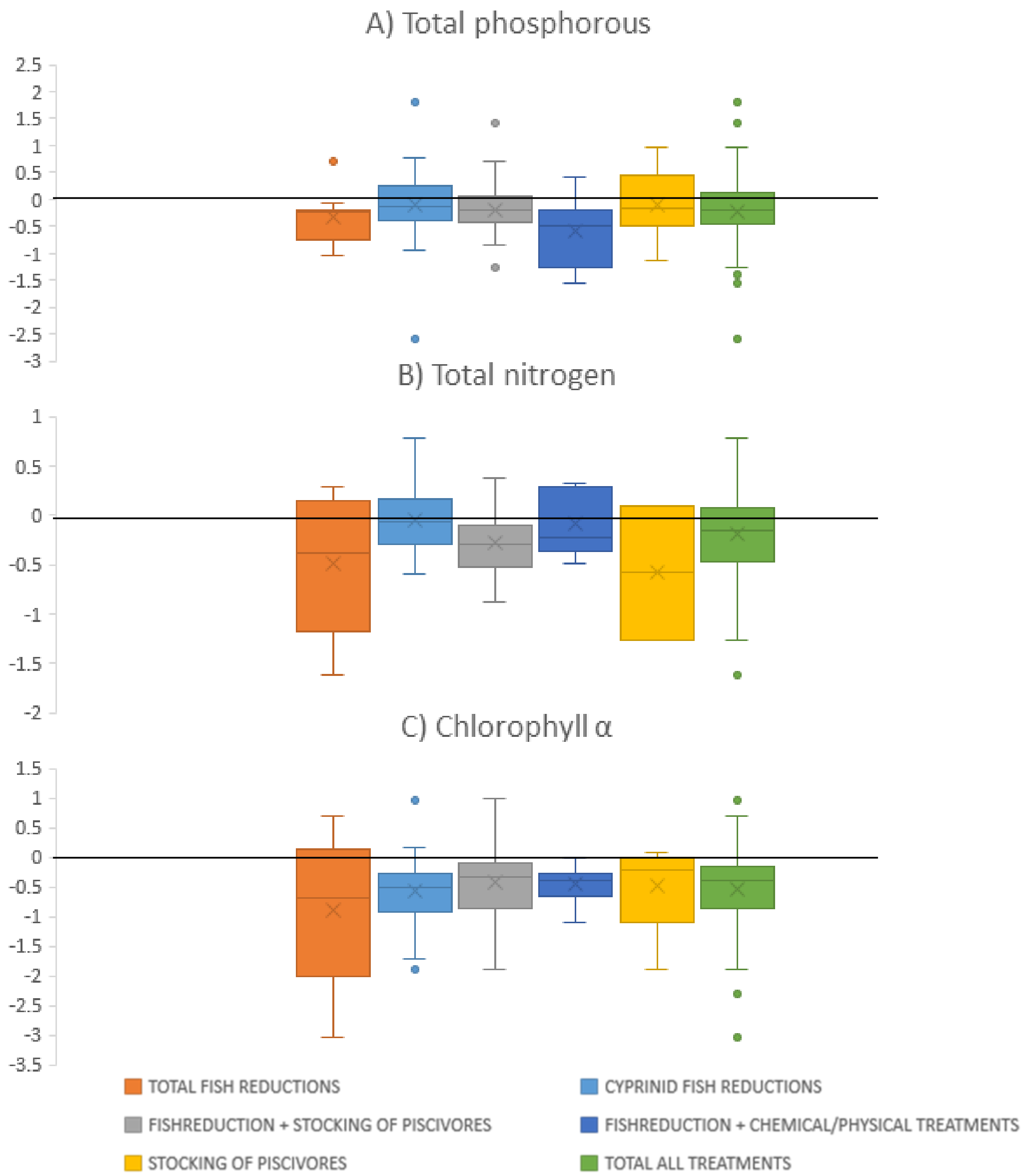

Figure 2: Effect size estimated as log response ratios (LRR) of A) Total phosphorus, B) Total nitrogen, and C) Chlorophyll α concentrations from before and after biomanipulations divided on different types of biomanipulations. 'Total all treatments' is when all treatments are combined. Boxes show 1st-3rd quantile, the bar the median and '×' the mean, whiskers indicate 1.5 times the inter-quantile distance, and dots outliers. The black line at zero indicates no change, whereas negative values a decrease in concentrations after treatments.

Cyprinid biomanipulations had a significant negative impact on chlorophyll α concentrations (LRR = -0.6, t_{25} = -5.2, $p < 0.001$, Fig. 2). This indicates that the phytoplankton abundance in these lakes decreased, despite that nutrient levels did not always decrease. The most noticeable positive response of the biomanipulations was the improvement in Secchi depth (Fig. 3), evident in around two thirds of the cyprinid biomanipulations. This may be related to a loss in resuspension of sediment particles when less cyprinids feed in the bottom sediment (Breukelaar et al. 1994; Adamek & Marsálek 2013). The distribution of submerged macrophytes and abundance of zooplankton were analysed in fewer studies, but in more than half of these there were an increase of macrophytes and zooplankton (Fig. 3). Studies investigating changes in fish conditions, i.e. growth in length or weight, in most cases reported an increase in the individual growth of the remaining fish, both for cyprinids and piscivores (Fig. 4).

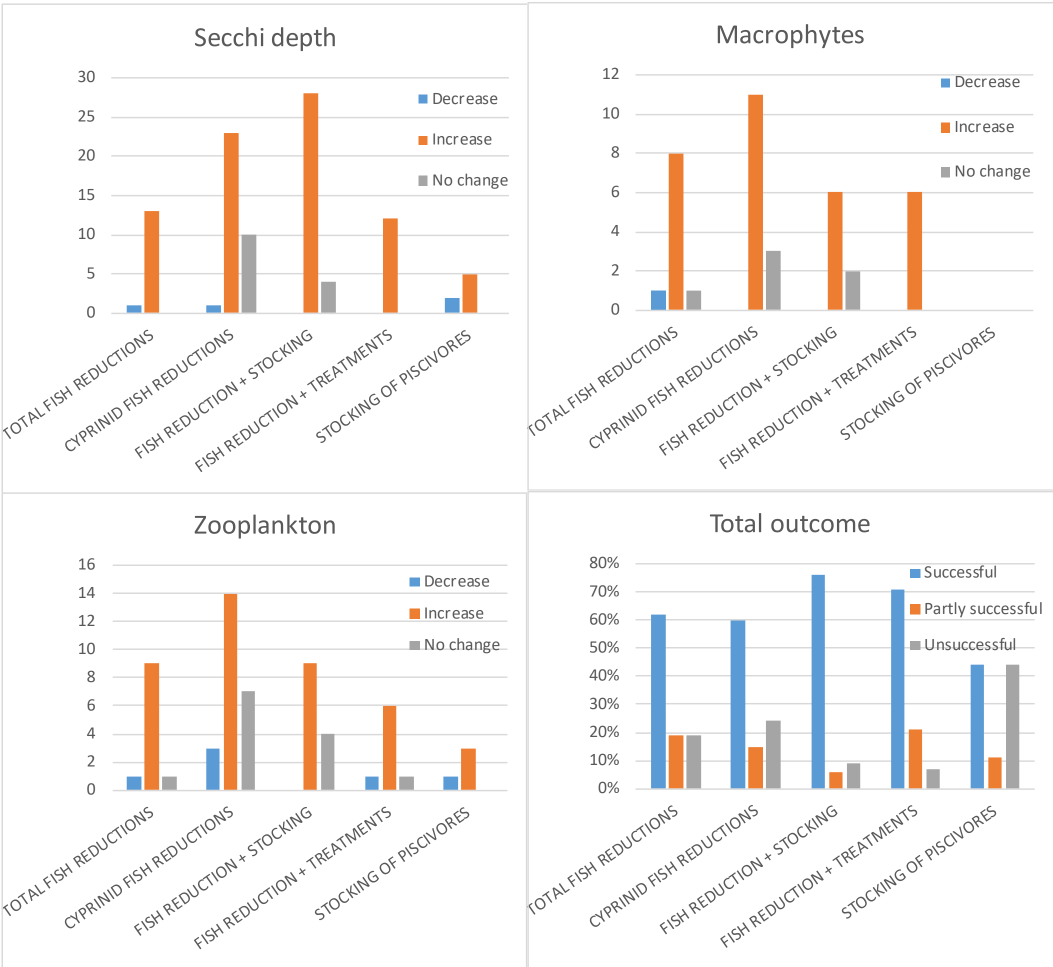

Figure 3: Number of lakes with changes in three different parameters (Secchi depth, Macrophyte cover/performance, Zooplankton abundance) following different biomanipulation treatments. Total outcome shows the proportion of all lakes with a specific biomanipulation treatment that was 'successful', partly successful and unsuccessful, see text for definitions.

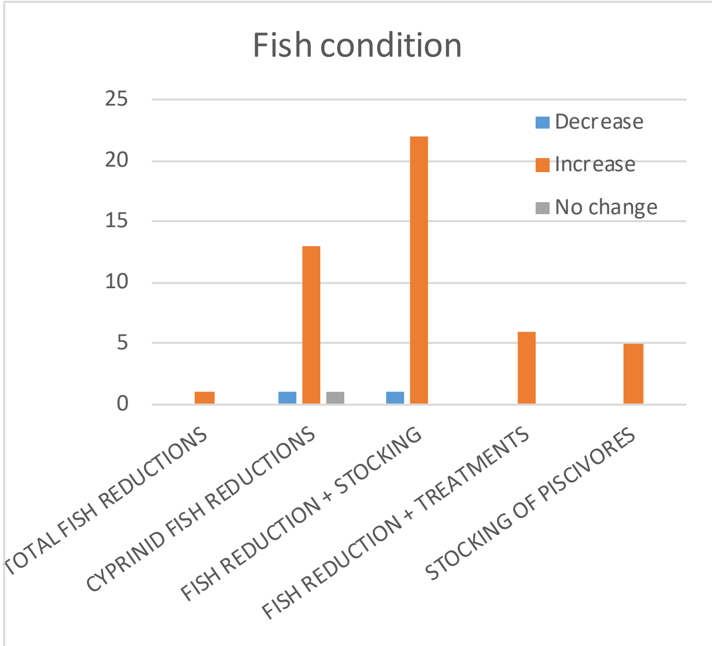

Figure 4: Number of lakes with changes in growth of length or weight for remaining fish following different biomanipulation treatments. Note that it applies both to remaining target fish (cyprinids) as well as piscivore fish.

3.1.1. Other biomanipulation treatments

A combination of cyprinid biomanipulation combined with additional nutrient reduction measures or stocking of piscivorous fish appears to be the most successful (93% success rate) strategy, with improvements in at least one of the water quality indicators (Fig. 3). Piscivore stocking only was the least successful method (56% of the studies was partly or fully successful, Fig. 3).

Overall, 65% of the 118 compiled studies successfully improved water quality by decreased phosphorus and/or chlorophyll α concentration and improved water transparency. This is in line with the study of Drenner and Hambright (1999), who estimated a 61% success rate for the 41 investigated biomanipulation experiments, and Hansson et al. (1998) who found that 59% of the 17 case studies reviewed were successful (measured by ≥15% decrease in phytoplankton biomass and turbidity). Søndergaard and colleagues (2007), who evaluated data from 70 restoration projects, found the strongest effects after 4-6 years after the start of fish removal, and a return to a turbid water state within 10 years or less in most cases, unless fish removal was repeated.

3.2. Biomanipulations in the Baltic Sea

In the Baltic Sea, this form of biomanipulation of cyprinids has not readily been employed as a restoration method, mainly due inherent difficulties in manipulating ecological processes on large spatial scales and in an open system (Lindegren et al. 2010; Appelberg et al. 2013). Some projects have tried to improve water quality in the coastal zone of the Baltic Sea using biomanipulation (Table 3).

Table 3: Fish reduction projects in the Baltic Sea, including the coastal area and country where the project was implemented, fishing methods (gillnetting (gn), fish trap (ft), seine fishing (se)), amount of the total fish catch and the average catch per year in tonnes, and the start and duration of the projects. Target species in Kalmarsund were herring (Clupea harengus) and sprat (Sprattus sprattus), while cyprinids were targeted in all other projects.

Coastal waters	*Country*	*Method*	*Total catch (tonnes)*	*Annual catch (tonnes)*	*Start (duration in years)*
Archipelago Sea[1]	FI	gn, ft	3555	444	2011 (8)
Kalmarsund[2]	SE	se	1500	750	2010 (2)
Kyrkviken, Gryt[3]	SE	se	8.1	8.1	2019 (1)
Mynälahti[4]	FI	gn, ft	255	128	2010 (2)
Pikkala Bay[5]	FI	ft	162	81	2009 (2)
Östhammarsfjärden[6]	SE	ft	7.3	4	2010 (2)

[1] Lappalainen et al. (2019); [2]Appelberg et al. (2013); [3]Klara Vatten Sverige AB (2019); [4]Setälä et al. (2012); [5]Jokinen & Reinikainen (2011); [6]Sandström (2011).

From the six examples of biomanipulations and targeted cyprinid fisheries in the Baltic Sea (Table 3) it is not possible to derive any general conclusions. Jokinen & Reinikainen (2011) tried to monitor ecosystem effects after fishing cyprinids for two years in the Gulf of Finland. They could not find any major effect of the fishery on nutrient levels, but there was a large variation in nutrient loads between years in the area. Despite the fishing targeting cyprinids, specifically bream, total fish biomass in the area had increased after the biomanipulation, mainly due to an increase in perch but also bream. It is not clear whether the increase of bream is part of the general increase of bream due to favourable conditions or an overcompensation of the species following intensive and targeted fishing (Jokinen & Reinikainen 2011).

Lappalainen et al. (2019) have evaluated fish stock effects from a targeted cyprinid fishery in the Finnish Archipelago Sea over eight years. This was done over a larger spatial scale, and large quantities of bream and roach were removed every year (200-450 tonnes per species and year). There were no effects on abundance (catch per unit effort) or size distributions of the targeted cyprinid fish species. The total instantaneous mortality rate for bream was estimated at 0.38 and approximately 0.4-0.6 for roach (Lappalainen et al. 2019). From the three Swedish examples of cyprinid-targeted fisheries there have been no evaluations of ecosystem effects or effects on fish stocks.

Assuming that 0.7% of the wet mass of bream is phosphorus and 2.5% nitrogen (Mäkinen 2008), a total of 35 kg and 50 kg fish-bound phosphorus in Kyrkviken and Östhammarsfjärden, respectively, and 133 kg and 181 kg nitrogen respectively was removed by these cyprinid fisheries.

3.2.1. Differences between freshwaters and marine waters

Hansson et al. (1998) argues that successful effects of cyprinid biomanipulations can only be expected when the external nutrient loading is less than 1.25 g m^2 per year. Further, significant changes in the ecosystem of shallow lakes may not occur unless the total phosphorus level is reduced below 0.075 mg P l^{-1} (Jeppesen et al. 2000), or 0.025 mg P l^{-1} in deep lakes (Sas 1989). The spatial variation in phosphorus levels in the Baltic Sea is large but in most coastal areas phosphorus is < 0.05 mg P l^{-1} (SMHI Vattenwebb, ICES 2019, Walve & Rolff 2020), and in theory, successful effects of cyprinid biomanipulations should be expected in most areas considering the experiences from biomanipulation in freshwaters. However, it is unknown if the same threshold levels for successful biomanipulation of cyprinids also applies to coastal conditions in the Baltic Sea. There is a substantial exchange of water masses in coastal areas (>50% per year in sheltered inner bays; SMHI Vattenwebb) with the open sea, which we judge will reduce the impact of local nutrient flows on local nutrient concentrations. Still, before initiating a targeted cyprinid fishery at a larger scale for improving ecosystem status in the Baltic Sea, we find it important to assess the nutrient loadings and concentrations in the specific area. In accordance with the predictions from lake studies, cyprinid fisheries may be more successful in reducing eutrophication symptoms in areas with lower net inflows and internal concentrations of phosphorus.

The chlorophyll α concentration in lakes before biomanipulation was on average 88 µg l^{-1}, which is generally much higher than in the Baltic Sea. In the most eutrophic coastal bays, chlorophyll α concentrations can reach 25-50 µg l^{-1} (Walve & Rolff 2020, SMHI Vattenwebb). This is similar to chlorophyll α concentrations even after successful biomanipulations in lakes (Table 2). Although cyprinid biomanipulations have been relatively successful in reducing chlorophyll α concentrations in lakes, a similar reduction may not be possible in coastal areas though an improvement may occur.

Few efforts have been made to estimate the biomass of fish in coastal areas of the Baltic Sea, but biomass will likely vary vastly between areas. The total biomass of non-piscivorous fish (likely dominated by cyprinids) was estimated to be 47-62 kg ha^{-1} in Kvädöfjärden Bay on the Swedish Baltic Sea coast (Bryhn et al. 2013). Adill and Andersson (2006) estimated the cyprinid biomass in Borholmsfjärden, south of Kvädöfjärden, to be 8.4 kg ha^{-1} bream and 3.2 kg ha^{-1} roach. However, these bays are not specifically eutrophic (Kvädöfjärden 0.025 mg P l^{-1}, SMHI 2013; Borholmsfjärden 0.22 mg P l^{-1}, Kenczek & Sunesson 2006), and a higher biomass of cyprinids is expected in shallow bays with higher nutrient concentrations.

In a pilot project the estimated biomass of cyprinids was ~165 kg ha^{-1} in Mynälahti Bay, Finland (Setälä et al. 2012; Table 3-4). In this area, around 8 kg ha^{-1} of cyprinids has been harvested yearly over 30 years (1980-2009) without evidently reducing the cyprinid populations. After two additional years of targeted fishing removing 16 kg ha^{-1} year^{-1} cyprinids, the stock was still estimated to be 159 kg ha^{-1}. The theoretical required annual catch for successful biomanipulation given the phosphorus levels would be 109 kg ha^{-1} (Jeppesen and Sammalkorpi 2002), based on calculations from lakes. Although these calculations are very rough, they indicate that cyprinid catches may need to be around 10 times larger than actual annual catches in the Baltic Sea. However, this calculation is based on experiences in lakes dominated by cyprinids with likely higher densities and testing is needed before applying it on the Baltic Sea conditions.

Table 4: Empirical and theoretical values of cyprinid catches from commercial fisheries in Mynälahti Bay, SW Finland, from 1980 to 2009 and in an intensive fishery period from 2010 to 2011, where the theoretical required catch for successful biomanipulation has been calculated according to the phosphorus concentration in the area (Jeppesen & Sammalkorpi 2002).The estimated cyprinid biomass before (1980-2009) and during (2010-2011) the intensive fishery (Setälä et al. 2012), and the actual and theoretical phosphorus reduction through removal of cyprinid biomass, calculated as 0.75% P content in cyprinid fish (Mäkinen 2008).

	Total P ($\mu g\ l^{-1}$)	Actual catch	Theoretical required catch	Estimated cyprinid biomass	P reduction through cyprinid removal actual/theoretical catch
Mynälahti Bay (1980-2009)	36*	8 kg/ha/yr 63 tonne/yr	109 kg/ha/yr 879 tonne/yr	165 kg/ha 1333 tonne	0.06/0.8 kg/ha/yr 0.5/6.5 kg/yr
Mynälahti Bay (2010-2011)	36*	16 kg/ha/yr 128 tonne/yr	109 kg/ha/yr 879 tonne/yr	159 kg/ha 1284 tonne	0.12/0.8 kg/ha/yr 1.0/6.5 kg/yr

**average long-term value (Setälä et al. 2012).*

Cyprinid fish contain about 0.7 to 0.8 % phosphorus depending on species (Mäkinen 2008), and the actual annual removal of phosphorus through the cyprinid fishery is estimated to be 0.5-1 tonne year^{-1} in the Mynälahti Bay. In a pilot study in Östhammarsfjärden, Sweden, Sandström (2011) found it reasonable to remove around 20 kg P ha^{-1} year^{-1} through targeted cyprinid fisheries. The estimated cost per unit of phosphorus removed was around the half of the estimated cost of the same amount of phosphorus removed by improved sewage water treatment.

Based on existing studies from lakes but also some coastal areas in the Baltic Sea, we conclude that realistic catches of a cyprinid fishery will have marginal local effects on phosphorus and chlorophyll α concentrations. However, a cyprinid fishery can still remove substantial amounts of nutrients from the Baltic Sea at a relatively low cost and might increase water clarity and macrophyte distributions. In the case of phosphorus removal, any fishery targeting cyprinids for food are required to report landings so official statistics can be used to estimate the amount of nutrients removed. To follow changes in water clarity and macrophyte distributions will require local environmental monitoring that is discussed more in detail in Chapter 5.1.

Although many biomanipulations of cyprinid fish in lakes have been successful, it cannot be assumed that similar projects will be successful in the Baltic Sea. Differences between lakes and the coastal zone with respect to size and openness are perhaps the most important features.

Lakes are more or less closed systems, whereas coastal systems are open, allowing fish populations to migrate in and out from different coastal areas (Geist & Hawkins 2016) and thus, perhaps reducing the impact of a potential targeted fishery. A tagging study for example showed that the migration distance of bream at the southern Finnish coast could be up to 300 km (Dahlström et al. 1968) although most tagged breams were only migrating 5 km or less.

Most biomanipulations of cyprinids in lakes are from lakes smaller than 50 ha and with an average depth of < 2 m (Bernes et al. 2015; Appendix A1). Coastal bays and lagoons are often comparable in size to lakes, but fisheries targeting (demersal) cyprinids may also be done in larger and deeper coastal areas. The larger volume of sea water means benthic habitats constitute a smaller reservoir of nutrients and fish biomass relative to the pelagic habitat. Removing demersal cyprinids in marine environments may therefore have a relative lower effect on the total nutrient pool compared to freshwater habitat.

Cyprinids in coastal waters and lakes may also differ in their diet (Rask 1989; Lappalainen et al. 2001) and habitat use (Geist & Hawkins 2016), which may affect the outcomes of biomanipulation. In lakes, rooted vegetation and insects are important food items (Geist & Hawkins 2016), whereas in the Baltic Sea cyprinids feed more on molluscs, like the blue mussel (*Mytilus edulis*) and the lagoon cockle (*Cerastoderma glaucum*) (Lappalainen et al. 2001; Rosenberg & Loo 1988). There can be a habitat differentiation between cyprinids of different sizes in the coastal area compared to lakes, since the spawning and nursery areas for fish at the coast are restricted to shallow and sheltered environments while the feeding areas of the larger adults are more widely distributed (Geist & Hawkins 2016).

Larvae and young cyprinids feed mainly on zooplankton. In the Baltic Sea, the zooplankton community is usually dominated by copepods, which are less effective in controlling phytoplankton compared to cladocerans that tend to dominate zooplankton communities in lakes (Meyer-Harms & Von Bodungen 1997; Engström-Öst & Al 2002). Thus, a reduction in cyprinid abundance may have a larger effect on phytoplankton abundance in lakes than in coastal areas due to different zooplankton communities.

3.3. Causes of limited success, failures and risks of biomanipulation

Aside from insufficient reduction of nutrients by biomanipulation compared to the extent of external phosphorus loading, the following mechanisms are proposed as explanations for failures or the limited durability of the effects of cyprinid biomanipulations in lakes (Søndergaard et al. 2007), that may also limit the ecosystem effects from a targeted cyprinid fishery in the Baltic Sea:

- Insufficient fish removal

A reduction in the biomass of cyprinids should be 75% or more for obtaining a clear water state in eutrophic lakes (Carpenter & Kitchell 1993; Hosper & Meijer 1993; Hansson et al. 1999). In almost all cyprinid biomanipulations in lakes, the percentage of cyprinids removed was an indicator of the projects' success. (Meijer et al. 1999). Such a high biomass removal (75%) is not likely from a targeted but sustainable cyprinid fishery in the Baltic Sea. It is currently not known how much can be sustainably harvested, but for other relatively long-lived species in the Baltic Sea, like cod and flatfishes, this is much less than 50% of biomass per year (ICES 2021).

- Recovery of zooplanktivorous fish

If cyprinid populations are heavily fished, the reduction can actually lead to a high recruitment of young-of-the-year (YOY) fish through overcompensation as a result of decreased competition between old and younger fish (Olin et al. 2006). As YOY cyprinids feed extensively on zooplankton (Romare & Bergman 1999), an increase of young cyprinids that in turn leads to an increase of the phytoplankton biomass can be a major reason for less successful biomanipulations in lakes (Hansson et al. 1998; Romare & Bergman 1999). However, if only the adult cyprinid stock is reduced by targeted cyprinid fisheries, an increased proportion of piscivore fish might be able to regulate the remaining cyprinid fish through improved predation control (Lammens 1999). In the Baltic Sea, which is more open for immigration at a local scale than a lake, cyprinid fish from neighbouring areas without a targeted cyprinid fishery could colonize the biomanipulated areas reducing the effect of targeted cyprinid fisheries.

- High wind resuspension

Sandy bottoms, conditions with prevailing wind and a high surface area to water depth might lead to constant turbid waters. This may pose a challenge for obtaining a more clear water state in shallow bays and lakes by the means of biomanipulation (Hosper & Meijer 1993; Benndorf 1995).

- Competitive release from other species and invasions

The amount of food for predatory fish may be reduced if cyprinid stock are reduced. Instead of promoting conditions for piscivorous fish, there is a risk that reduced competition from cyprinid fish favour other competing species with a similar diet. In the Baltic Sea for example, the three-spined stickleback (*Gasterosteus aculeatus*) has increased during last decades (Bergström et al. 2015; Olsson et al. 2019) and might be favoured by lowered competition for food with cyprinid fish. Sticklebacks feed on eggs and larvae of piscivorous fish and may therefore prevent recovery of piscivorous fish (Olsson et al. 2019).

There is also a risk for expansion of invasive species like the round goby (*Neogobius melanostomus*) if there is decreased competition from cyprinids. Additionally, non-vertebrate competitors with cyprinid fish such as *Saduria entomon*, could replace cyprinids, but as they are food for fish the net effect is difficult to predict. Invertebrate predators such as the mysid crustacean (*Neomysis*) or predatory water fleas (*Leptodora*) can also invade an area and prey on zooplankton, which in turn may result in increased phytoplankton abundance (Hosper & Meijer 1993). Another risk with a targeted cyprinid fishery might be that for example comb jellies (*Mnemiopsis sp.*) adopt the food niche of zooplanktivorous fish (Hansson 2008). Similar changes have been seen in the Black Sea, where after problems with eutrophication and overfishing, the warty comb jelly (*Mnemiopsis leidy*) increased explosively when the biomass of planktivorous fish collapsed (Daskalov et al. 2007). *Mnemiopsis* benefits from turbid water and high concentration of zooplankton. Comb jellies are, however, strongly salt-limited in the Baltic Sea (Jaspers et al. 2011) but this example illustrates that when the structure of ecosystems changes radically, they do not necessarily return to the original state.

- Inedible phytoplankton

At high nutrient concentrations there is an increased chance of high abundance of inedible cyanobacteria (Benndorf 1995). Hence, although there might be an increase in zooplankton due to cyprinid fisheries, the zooplankton cannot feed or consume the phytoplankton if these are of low quality or even toxic.

4. Contaminants

4.1. Contaminants in cyprinids

For fish to be used as human food it is important that fishing is sustainable and does not risk the health of the consumers. There are several examples of fish species from the Baltic Sea and Swedish lakes contaminated with pollutants at levels that require restrictions in human intake (Swedish Food Agency 2020). Dioxins, PCBs (Polychlorinated biphenyls), PFASs (Perfluorinated alkylated substances) and mercury are contaminants of particular relevance when it comes to human consumption of Swedish fish (Swedish Food Agency 2020). In addition, brominated flame retardants (PBDEs) have been detected in fish at levels that require caution (Swedish Food Agency 2017). The contaminants mentioned above are also identified substances among the 45 priority substances listed in the European Water Framework Directive (The European Parliament 2013).

The Swedish Environmental Research Institute (IVL) examined levels of contaminants (mercury, dioxins, PCBs, PFOS and PFOA) in bream from mainly the large lakes in Sweden. Only PFOS in bream from Lake Vänern exceeded the environmental quality standard for biota, whereas mercury, PCBs and dioxins were well below the European maximum levels for human consumption (Waldetoft & Karlsson 2020). In a study conducted on bream in Finjasjön a wide range of contaminants were analysed (Annadotter et al. 2019). The concentrations of PFOS and PAHs indicated bream in Finjasjön were exposed to elevated levels of these contaminants but safe to consume at least once a week.

In this study we focused on mercury, cadmium, dioxins, PCBs, PFASs, and brominated (PBDEs) flame-retardants. Due to their hazardousness and spread in the environment, these substances were assessed to be the most relevant for initial estimation of risks for exposure of contaminants due to consumption of cyprinids.. Dioxins, PCBs and PBDEs tend to accumulate in fat rich tissues, whereas PFASs, mercury and cadmium does not. Hence, consumption of leaner fish species does not necessarily correspond to a lower intake of PFAS and heavy metals. Common for all the assessed substances is however that they tend to magnify higher up in the food chain. Bream has a relatively low fat-content, compared to many other fish, which might explain the low contaminant levels so far found for bream in Swedish lakes (Annadotter et al. 2019, Waldetoft & Karlsson 2020). Nonetheless, there is a gap in our knowledge on levels of contaminants in bream, ide and roach.

4.1.1. Threshold levels

There are several types of threshold levels for contaminants in food and biota (fish) (Table 5). The European maximum allowed level (MAL) is the proportion of wet weight of a contaminant that is safe to consume according to the European Food Safety Administration (EFSA; European Commission 2006). The Tolerable Weekly Intake (TWI) is a measure decided by EFSA that indicates the cumulative dose of contaminant per kilogram body weight of a person, which is safe to eat every week throughout life. The Reference Dose (RfD) is similar to TWI, indicating the daily intake level at which an accumulated intake over a lifetime does not pose intolerable risks to human health (Lychee et al. 2015). The Reference Dose is a threshold value used in USA and has therefore only been used when no corresponding threshold value has been available for Europe. The estimated weekly consumption (C_{week}) of a fish species (throughout life) that would be safe regarding intake of contaminants is calculated as:

$$C_{week}\ (\mathrm{g}) = TWI(\mathrm{g/kg}) * bw(\mathrm{kg}) / aveConc(\mathrm{g/g\ ww})$$

where *bw* is a standardized bodyweight (here 60 kg), and *aveConc* is the average concentration of a contaminant in wet weight of respective fish species.

In addition, there are defined thresholds for concentrations in biota. The European Environmental Quality Standard (EQS) indicates threshold levels for risk of harm to biota and ecosystems (European Parliament 2013). The EQS-values do not take risks for human health into account, but can be used as a pointer when threshold values for human health are not available. All types of threshold levels are not available for all contaminants.

Table 5: Overview of different threshold levels for the analysed contaminants. MAL is Maximum allowed level in food (concentration in wet weight of fish muscle), TWI is Tolerable weekly intake in from all sources (total weight per week and kilo bodyweight), RfD is Reference dose from all sources (total weight per day and kilo bodyweight), EQS is Environmental Quality Standard in biota (fish) (concentration in wet weight of fish muscle). Ave. Concentration is the estimated upper bound concentration from all analysed samples.

Contaminant	MAL	TWI	RfD	EQS	Ave. Concent-ration
Mercury	0.5 mg/kg	0.0013 mg/kg	NA	0.02 mg/kg	0.12 mg/kg
Cadmium	0.05 mg/kg	NA	NA	NA	0.001 mg/ kg
Dioxin	3.5 pg TEQ/g	NA	NA	NA	0.45 pg/g
Dioxin and dioxin-like PCBs	3.5 pg TEQ/g	2.0 pg TEQ	NA	6.5 pg TEQ/g	0.75 pg TEQ/g
(Non dioxin-like) PCBs	125 ng/g	NA	NA	NA	1.96 ng/g
PFASs	NA	4.4* ng/g	NA	9.1 ng/g	1.05 ng/g
PBDE	NA	NA	7.4** µg/kg	0.0085 µg/kg	0.46 µg/kg

PFOA+PFOS+PFNA+PFHxS, See Table 8 for conger-specific levels*

4.1.2. Mercury and cadmium

Mercury is both a naturally occurring element and an environmental pollutant from anthropogenic sources such as mining, burning of coal and waste, as well as from various industrial activities. It is present in the environment as either elemental mercury, ionic mercury or organic (methyl- or ethyl) mercury. Mercury is persistent in the environment and methylated mercury bio-accumulates and biomagnifies up the food-web (WHO 2008).

Human exposure to mercury is associated with a variety of harmful effects. Exposure to inorganic mercury is associated with damaging effects on the kidneys. Both elemental mercury and methylmercury interfere with the nervous systems (central- and periphery), especially under early developmental stages. Thus, children and foetuses are vulnerable to exposure of mercury (WHO 2008). The main source of exposure to mercury is via intake of fish, both globally (UN Environment, 2019) and in Sweden (Swedish Food Agency 2017).

Cadmium naturally occurs in soils, but is also spread through human activities, for instance, dispersal of industrial fertilizers or bio-sludge on farmland. Another source of cadmium is atmospheric deposition from combustion of fossil fuels and waste, and historically also from metal smelters. Cadmium affects the kidneys and is classified as carcinogenic to humans by the International Agency for Research on Cancer. For smokers, the dominant source of exposure is through cigarettes. For non-smokers the main source of exposure is via food. In Europe, the exposure from food is generally below the recommended threshold levels, but the accumulating properties of cadmium is a reason to limit exposure nonetheless. The main food items containing cadmium are liver, kidneys, shellfish and certain kinds of mushrooms (EFSA 2012).

4.1.3. PCBs and dioxins

Polychlorinated biphenyls (PCBs) are a group of chemicals consisting of 209 different congeners. These were formerly used in various industrial processes, for instance in transformers and sealants used in buildings. In 1978 the use of PCBs in new products was prohibited in Sweden. In 1995 the prohibition was expanded to all products.

Dioxins are a large group of chemicals consisting of the two sub-groups polychlorinated dibenzo-p-dioxins (PCDDs, 75 congeners) and dibenzofurans (PCDFs, 135 congeners). They are mainly unintentional by-products of various industrial processes and combustion processes, but volcanic eruptions and forest fires are also sources of dioxins. Both PCBs and dioxins are considered to be persistent organic pollutants (POPs). This means that they are very resistant to degradation, they bioaccumulate and they have adverse effects on human health and the environment. Acute toxic effects from dioxin exposure includes permanent skin lesions (known as chloracne) and impairments of the liver function. Effects from long-term exposure to dioxins include neurotoxic- and endocrine-disruptive effects as well as impairments of the immune system and reproductive functions. Fetuses are extra vulnerable to exposure. PCBs with a planar molecular structure (so called dioxin-like PCBs) have similar toxicological properties and effects as dioxins. These are therefore usually considered together in contexts dealing with human health and risk assessment (WHO 2010). Non-dioxin-like PCBs are associated with negative effects on the liver, the endocrine and the immune system.

Dioxins and PCBs are commonly present in fatty fish from the Baltic Sea and Lake Vänern and Vättern in concentrations exceeding the maximum levels allowed in food by the European Commission Regulation (EC No 1881/2006). Sweden, however, has been granted an exemption to these maximum levels, providing that the population is properly informed and provided with guidelines on maximum intake of fish from the affected areas (European Commission 2011).

4.1.4. PFASs

Perfluorinated alkylated substances (PFASs) are a group of approximately 4700 different substances, used for a wide variety of purposes. Among these, PFOS (Perfluorooctanesulfonic acid) and PFOA (Perfluorooctanoic acid) are the most well known. Many PFAS have excellent water-repellent properties and are used in non-stick coatings, coatings for packaging materials and waterproofing of textile materials. Another important source of PFAS are firefighting foams, which has been the cause of several cases of pollution of groundwater resources in Sweden. For areas with non-polluted drinking water, fish are the most common source of exposure to PFAS in Sweden (Livsmedelsverket 2016). The various kinds of PFASs have different properties, but they are all extremely resistant to degradation and can easily spread in the environment. PFAS-molecules with a long carbon chain have shown to bio-accumulate, in particular in fish.

Much is unknown regarding the whole group of substances, but for a few (for example PFOS and PFOA) there is evidence of harmful effects on human health. Among the effects are endocrine disruption, lowered functionality in the liver and inhibitory effects in foetal development (EEA 2020).

4.1.5. PBDEs

Polybrominated diphenyl ethers (PBDEs) are a group of substances used as flame-retardants in a variety of different materials such as furniture, electronics and plastics. There are 209 different congeners of PBDEs and they are widely spread throughout the environment. The properties of the different congeners vary, but in general, PBDEs are persistent and bioaccumulate. The main source of exposure to humans is via intake of fish. PBDEs can have negative effects on the development of the nervous system (EFSA 2011).

4.2. Sampling, preparation and calculations of concentrations

Samples of bream, ide and roach were obtained through commercial fishers from five different locations along the Swedish coast of the Baltic Sea during autumn 2020 (Figure 5). At one of these sites, Norrsundet, there was an active paper-mill until 2007, where the fibre-banks in the adjacent sea have high levels of organic toxins like dioxins and of cadmium and mercury (Norrlin et al. 2016).

The species and number of individuals collected varied between locations (Table 6). We measured total length, to closest cm (Appendix 2), and weight of each sampled fish. All fish were of sizes for commercial interest. For all fish a muscle sample was examined for contaminant content, as it is only the muscle that will be used for human consumption. However, in comparison with other studies, whole fish can be used, which includes brain and intestines that may have higher concentrations of contaminants due to more fat-rich tissue.

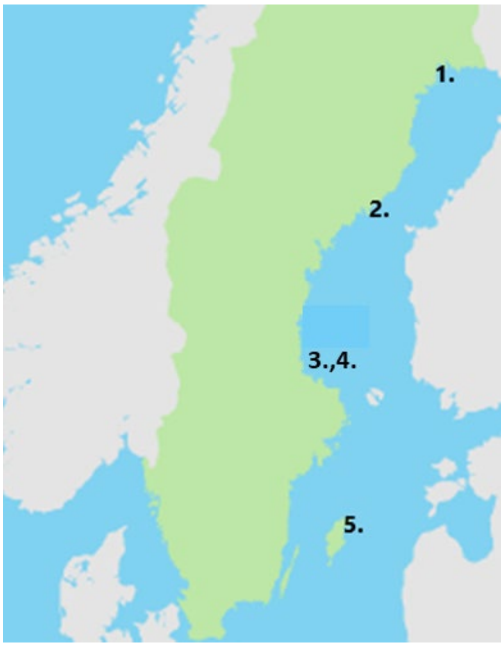

Figure 5: Locations of sites of capture. 1: Kalix Archipelago, 2: Husum, 3: Norrsundet, 4: Sörsundet, 5: Herrvik.

Table 6: Number of fish collected from each sampling site and species. Numbers within brackets are size range of fish. See Appendix 2 for length and weight of each sampled fish.

	Bream	**Ide**	**Roach**
Husum	-	14 (39-47 cm)	-
Kalix Archipelago	21 (26-48 cm)	6 (41-46 cm)	29 (21-31 cm)
Norrsundet	24 (21-55 cm)	-	14 (20-31 cm)
Herrvik	-	11 (27-52 cm)	-
Sörsundet	-	-	6 (20-25 cm)

For the analysis of contaminants, aggregate samples of fish muscles were prepared in duplicates for each species and sampling site. Most samples were aggregates of 10 different individuals, but in some cases due to few available fish an aggregated sample could consist of 2-5 individuals (e.g. ide from Kalix and Herrvik and roach from Sörsundet; Table 6; Appendix 2).

For the analysis of mercury, and cadmium, the aggregate samples were prepared in duplicates in a range between 5-10 g. For the analysis of dioxins and PCBs the aggregate samples ranged between 28-90 g. Aggregate samples for PFAS analysis ranged between 3-6 g per duplicate and sampling site. PBDE samples ranged between 6-29 g. Due to limitations in availability of fish, only three aggregate samples, from two different sites (Kalix and Herrvik), could be analysed for PBDE.

The analysis of PFAS was performed by the Organic Risk Pollutants Laboratory at the Department of Aquatic Sciences and Assessment, Swedish Agricultural University in Uppsala. The analysis of mercury, cadmium, dioxins, PCBs and PBDEs were performed by ALS Scandinavia. Both Laboratories are accredited, for the analyses in question, by Swedac (https://www.swedac.se/).

For mercury, there is a relationship between the size of the fish and the concentrations of mercury in the tissue (Meili et al. 2004; Schütze et al. 2004; Sundbom et al. 2006). To compare between sites and species to adjust for differences in size, we calculate the weight-normalized mercury concentrations Hg_{NW} according to:

$$Hg_{NW} = Hg_F/(a \cdot W^b + c)$$

where Hg_F is measured concentration in an aggregated sample, a is a slope-factor depending on species, W is average wet weight of fish in the aggregated sample, b is a weight factor that has been estimated to 2/3 for any fish species, and c represents the corresponding levels in gonads and has been estimated to 0.13 for any species (Sundbom et al. 2006). Hence, Hg_{NW} estimates the expected mercury concentration of a fish weighing one kilo.

According to Sundbom et al. (2006), a = 1.0 for roach and we used this value for the normalization of mercury concentration for all species. There are no available values of a for bream and ide and the standardization of fish body mass for these species was done using a simpler method where the mass of the sampled fish was standardized to a fish at 1kg, i.e. dividing the mercury concentration with the fish weight (Waldetoft & Karlsson 2020).

For each contaminant we calculate the average and standard deviation of concentration across both all aggregated samples (all three species: n = 17) and for each species separately (ide: n = 6; bream: n = 5; roach: n = 6, except for PBDE that was calculated only on three samples of ide) (Appendix 3-7). For each contaminant, we calculated the both average lower and upper bound concentrations. Average lower bound concentration is the average concentration of measured concentrations. This assumes samples with concentration below the detection threshold had zero levels of a contaminant. It is not likely true but gives a lower bound of average concentration. Average upper bound concentration is the average of observed values and the detection limits for concentrations below the detection limit. This account for concentrations that may be too low to be detected but may still be present. Thus, the upper boundary and represents a “worst case” concentration. The true concentration will most likely be somewhere between lower and upper bound concentrations. For example, if three samples have measured concentrations 0, 0.5 and 1, and the detection threshold is 0.2, then the average lower bound concentration is 0.5 and the average upper bound 0.7 and we can assume the true average concentration is in the range 0.5-0.7.

4.3. Contaminants: Results and Discussion

4.3.1. Mercury and cadmium

MAL for mercury in food is 0.5 mg/kg wet weight fish muscle (European Commission 2006). EQS for mercury in biota is 0.02 mg/kg wet weight fish muscle (European Parliament 2013).

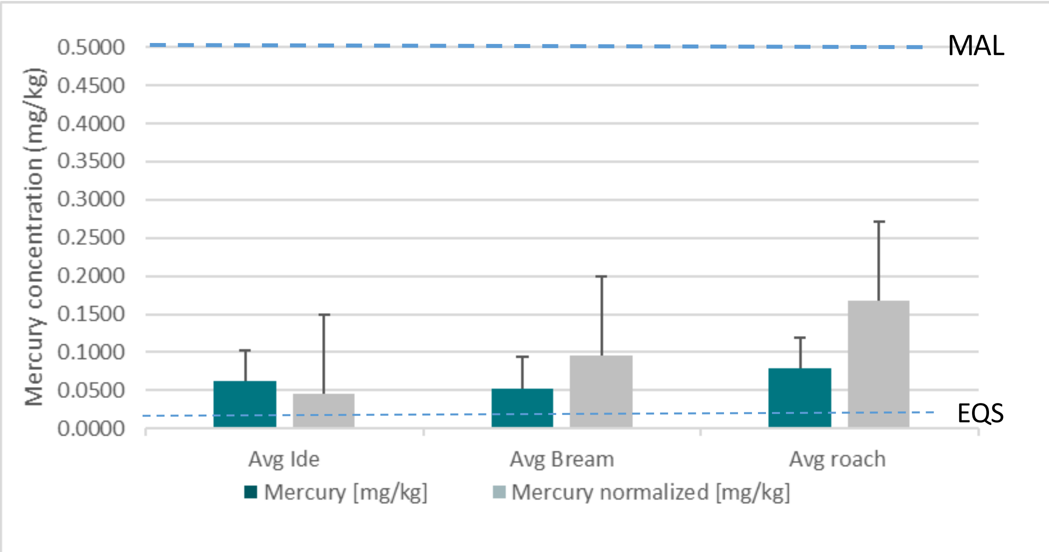

Figure 6: Mean measured and (weight) normalized concentrations of mercury (mg/kg wet weight) in aggregated samples of ide, bream and roach, respectively. Error bars show standard deviation across aggregated samples. MAL shows threshold value for Maximum Allowed Level and EQS for Environmental Quality Standard.

Bream from Kalix had twice the mercury concentration compared to Norrsundet (0.08 vs. 0.04 mg/normalised kg), whereas the opposite was found for roach (0.14 vs 0.35 mg/normalised kg) and differences between sites were small for ide (0.04-0.06 mg/normalised kg) (Figure 6, Appendix 3).

All standardized and non-standardized levels of mercury are well below the MAL but surpassed the EQS of 0.02 mg/kg in fish muscle (Table 5, Fig. 6). Due to the widespread distribution of mercury, from mainly atmospheric deposition, levels above EQS are ubiquitous in fish caught in Sweden (VISS: https://viss.lansstyrelsen.se/; Sorensen & Faxneld, 2020).

TWI of mercury established by EFSA (2012) is 0.0013 mg/kg bodyweight. Based on the average standardized concentrations of mercury (Fig. 6), for an adult weighing 60 kg the TWI would be reached at a consumption level of 1.56 kg of ide, 0.78 kg of bream or 0.34 kg of roach per week.

For cadmium, MAL is 0.05 mg/kg wet weight (European Commission 2006) and there is currently no EQS for cadmium in biota. All samples except one were below the detection level of 0.001 mg/kg (Appendix 3).

4.3.2. Dioxins and dioxin-like PCBs

MAL of dioxins in fish is 3.5 TEQ (toxic equivalency) pg/g ww (wet weight), and for the sum of all dioxins and dioxin-like PCBs in fish is 6.5 pg TEQ/g ww (European Commission 2011). The average estimated concentrations of dioxins in the cyprinid samples were all below 3.5 pg/g ww and only two of the 17 samples had concentrations above the limit of quantification (Fig. 7; Appendix 4). The two samples above the limit of quantification (0.94 pg/g ww, Appendix 4) were bream from Norrsundet which was not unexpected as there are contaminated fibre-banks in the area (Norrlin et al. 2016), but yet below the MAL of 3.5 pg/g.

Ide was the species having highest sum of dioxins and dioxin-like PCBs (sum WHO-PCDD/F- PCB-TEQ), 1.3 pg/g ww (Fig. 7). Still, all samples were far below the MAL of 6.5 pg/g ww in fish for human consumption (Fig. 7, Appendix 4).

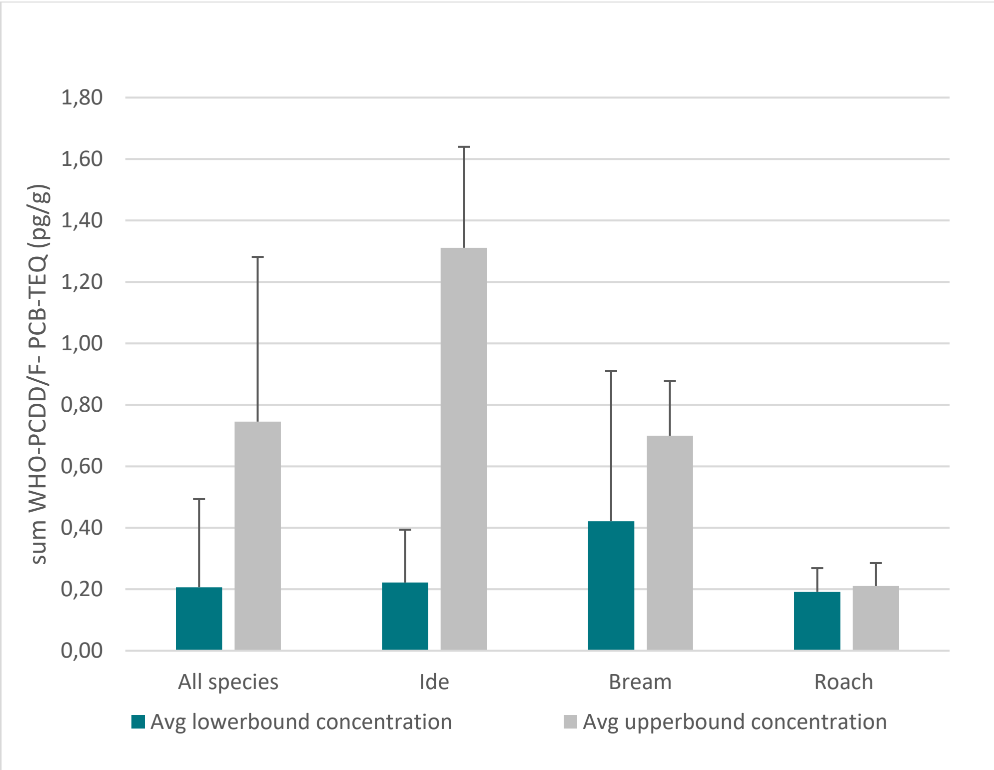

Figure 7: Mean measured (lower bound) and worst case (upper bound) concentrations of the sum of dioxins and dioxin-like PCBs (WHO-PCDD/F- PCB-TEQ) in aggregated cyprinid fish from the Baltic Sea. Error bars are standard deviation among aggregated samples.

Observed levels in cyprinids were low compared to average toxic equivalents (TEQ) of dioxins and dioxin-like PCBs in herring from the Baltic Sea, 4.2 pg TEQ/g ww, and 9.4 pg TEQ/g ww for herring caught in the Gulf of Bothnia (National Food

Agency Sweden 2013). Cyprinids have less fat content (0.5-2.0% fat in our samples, Appendix 3) compared to herring (3.5-10.4%, Aro et al. 2000, Szlinder-Richert et al. 2010), and therefore probably also a lower content of healthy fatty acids. The results indicate that it is possible to consume cyprinids more often than herring, without being exposed to levels of dioxins and dioxin-like PCBs that may pose a risk to human health.

The tolerable weekly intake (TWI) of dioxins and dioxin-like PCBs is 2.0 pg TEQ per kg bodyweight (EFSA 2018). Using the higher upper-bound concentration as a worst-case scenario, an adult of 60 kg can safely consume 0.092 kg of ide, 0.172 kg of bream or 0.57 kg of roach per week.

4.3.3. Non-dioxin like PCBs

For the sum of six indicator congeners (PCB6-ICES) of non-dioxin-like PCBs MAL in fish is 125 ng/g ww in freshwater fish and 75 ng/g ww in marine fish (European Commission 2011). Even though the levels of (non-dioxin) PCB (ICES-6) in bream and roach from Norrsundet was much higher (3.75 ng/g ww; Appendix 5) than average (1.47 ng/g; Fig. 8), again all levels were well below the MAL of 75 ng/g ww for marine fish. To date there is no TWI available for non-dioxin like PCBs and thus, no estimation of safe intake was possible.

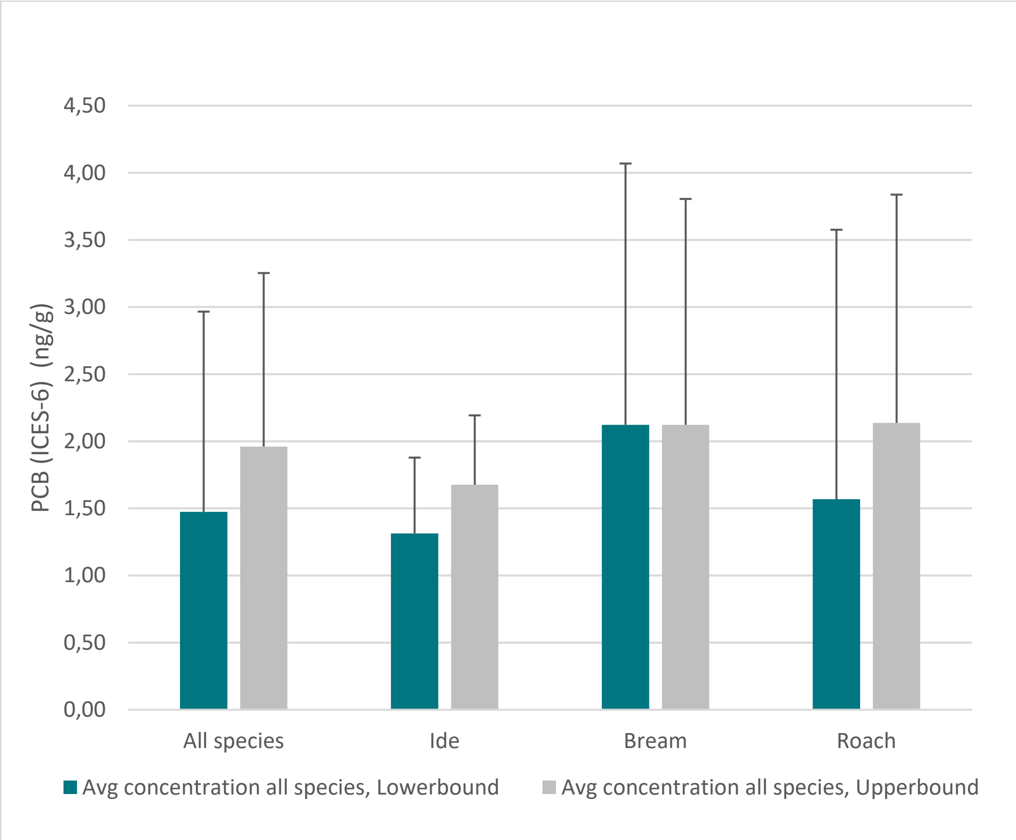

Figure 8: Mean (± standard deviation) measured (lower bound) and worst case (upper bound) concentrations of PCB (ICES-6) of aggregated samples of cyprinid fish from the Baltic Sea. Error bars are standard deviation among aggregated samples.

4.3.4. PFAS and PFOS

The majority of samples had concentrations of PFAS below the level of quantification, but some samples of bream and ide showed concentrations well above the detection limit (Fig. 9; Appendix 6). The samples with detected levels of PFASs came from three sampling sites: Kalix, Norrsundet and Herrvik (Appendix 9). Note that only one of the two aggregated samples from each site was above the detection limit. That one aggregate sample shows considerably higher concentration than the other from the same sampling location indicates substantial variation in concentration between individuals or that concentrations are close to detection.

The European Union has not decided on any MAL for PFAS, but the established EQS for PFOS in biota is 9.1 ng/g wet weight fish muscle (The European Parliament 2013). All samples of cyprinid fish from the Baltic Sea were far below the EQS for PFOS (Fig. 7). The average upper bound concentrations (Fig. 9) of PFOS were 0.40 ng/g (ide) and 0.87 ng/g (bream). PFOS was not detected in roach and upper bound concentration therefore 0.19 ng/g.

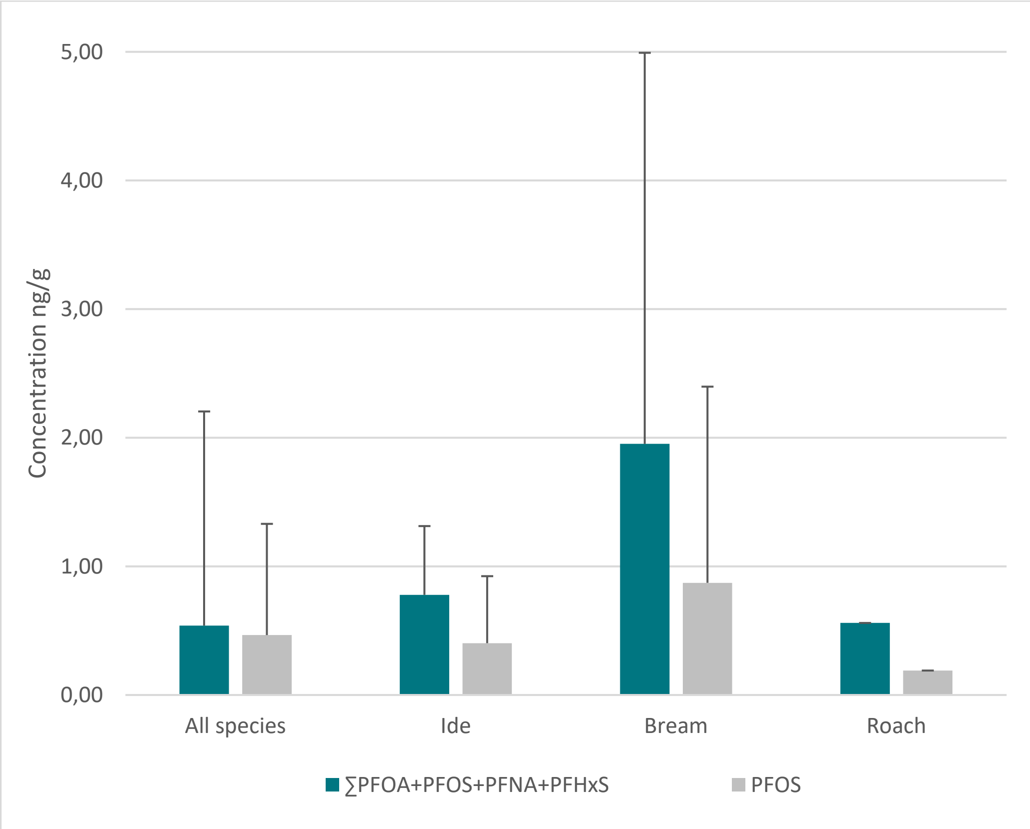

Figure 9: Mean upper bound ("worst case") concentrations of PFOS only and the sum (∑) of PFOA, PFOS, PFNA, PFHxS in ide, bream and roach from the Baltic Sea. Error-bars show the standard deviation between aggregated samples.

For most of these congeners, MAL, EQSs or TWIs are not yet available. However, knowledge about PFASs is under development, and these congeners may be of interest to monitor for the future (EFSA 2020). The TWI for the sum of the four substances of PFOA, PFOS, PFNA, PFHxS is 4.4 ng/kg bodyweight (EFSA 2020), and the upper bound concentrations estimated here (Fig. 7) were 0.78 ng/g for ide, 1.93 ng/g bream and 0.58 ng/g roach. For an adult of 60 kg the TWI would be reached on average at a consumption of 0.338 kg of ide or 0.138 kg of bream per week. Based on the concentration of the single highest sample of bream (7.3 ng/g, Appendix 6) it would be safe to consume 36 g bream per week, whereas for the samples with non-detected concentrations (upper bound level 0.56 ng/g) TWI would be reached at a consumption of 471 g per week. Hence, there is a substantial variation between samples in what amount that is considered to be safe to consume. As there is no evident differences between species or sites, a common intake rate for all cyprinid species could be relevant. The upperbound average of PFOA, PFOS, PFNA, PFHxS for all three species was 1.05 ng/g, which corresponds to an intake of fish at 251 g/week that is considered as safe to consume.

4.3.5. PBDEs

Detectable levels of PBDEs in ide were only found in the two Herrvik samples (Table 7). There are no MAL or TWI established by the European Union but the EQS for the sum of PBDE 28, 47, 99, 100, 153 and 154 is 0.0085 µg/kg wet weight fish muscle (EFSA 2011). The measured concentrations of PBDE in ide were well above the EQS and the main contributor in ide was BDE 47 (Table 7). However, all other measured fish sampled from the Baltic Sea exceed EQS for PBDEs, being in the range 0.03-0.82 µg/kg (Helcom 2018b). In fact, EQS is so low relative detection limits for standard analyses (used here) that the upper boundary exceeds (0.4 µg/kg) the EQS almost 50 times.

Table 7: Concentrations of BDE 47 and 99, and the sum of BDE 28, 47, 99, 100, 153, and 154 in ide from Herrvik and Kalix archipelago. Note that the upper bound concentration at Kalix is higher than those from Herrvik due to levels are from the limit of quantification (0.10 µg/kg), while levels from Herrvik we used the limit of detection at 0.05 µg/kg as other PBDE were detected. Therefore, we can use the more stringent limit of detection for the ide from Herrvik.

Species	Site of capture	Avg mass of individuals in aggregate sample	BDE 47	BDE 99	Sum of BDE lower bound (28, 47, 99, 100, 153, 154)	Sum of BDE upper bound (28, 47, 99, 100, 153, 154)
		kg	*µg/kg*	*µg/kg*	*µg/kg*	*µg/kg*
Ide	Kalix	1.030	<0.10	<0.1	0	**0.60**
Ide	Herrvik	1.625	0.11	<0.05	**0.11**	**0.36**
Ide	Herrvik	1.819	0.17	<0.05	**0.17**	**0.42**
	Mean		0.13	0.07	**0.14**	**0.46**

For the congeners BDE 47, BDE 99, BDE153, and BDE 209 there is a health-based target value, Reference-dose (RfD), established by the US EPA (Table 8). Based on the average concentrations in Table 7, an adult would be able to eat 323 kg of ide per week before surpassing the RfD for BDE 47, and 600 kg of ide before surpassing the RfD for BDE 99.

Table 8: Reference doses (RfD) for four different congeners of PBDE, estimated by US EPA (US EPA 2017).

Congener	RfD
BDE 47	0.1 µg/kg/day
BDE 99	0.1 µg/kg/day
BDE 153	0.2 µg/kg/day
BDE 209	7.0 µg/kg/day

5. Monitoring of ecosystem effects and contaminants

Biomanipulation of cyprinids in lakes aims to restore or improve water quality, with less emphasis on using cyprinids as a food resource. Although short-term biomanipulations of cyprinids have been conducted in the Baltic Sea, and may increase in the future, the targeted cyprinid fishery in the Baltic Sea is currently aiming towards sustainable human food production, with potential positive side effects on water quality. Cyprinid fisheries in the Baltic Sea may therefore have longer term changes in cyprinid stocks and food-webs compared to the short-term complete removal of cyprinids that is desired for in lakes. As a result, marine cyprinid fisheries will require longer term monitoring of ecosystem effects. In this section, we list and prioritize monitoring of different ecosystem effects and contaminants in conjunction with a targeted cyprinid fishery in the Baltic Sea (Table 9). Sundblad et al. (2021) have previously reviewed different metrics to be used in stock specific monitoring for the exploited cyprinid stocks.

Table 9: Overview of monitoring activities and priorities in relation to evaluations of a targeted cyprinid fishery at the Baltic Sea.

Activity/ Monitoring	Goal/aim	Priority	Frequency	Comment
Fish	Fulfill MSFD* targets for fish	Medium-high	Every to third year	Costly, can be done in collaboration with cyprinid fisheries
Zooplankton	Increase abundance	Low	2-4/y	Costly
Zoobenthos	Increase abundance	Low	1-2/y	Costly
Phytoplankton	Decrease abundance	Low	2-4/y	Costly
Chlorophyll α	Decrease concentration	High	3-10/y	*In situ* device (cheaper) or lab analysis (more accurate)
Submerged macrophytes	Increase abundance	Medium-high	2^{nd}-4^{th}y	Long term changes
Water transparency	Increased transparency	High	3-10/y	Secchi depth or light attenuation
Nutrient concentrations	Decreased concentration	Low-Medium	1/y	Total phosphorus and nitrogen most relevant
Oxygen concentrations	Increased concentration	Low-Medium	2-4/y	Mainly in areas with known historic hypoxic condition
Contaminants	Below threshold values	Medium-high	5^{th}y	Dioxins/dioxin-like PCBs and PFAS most prioritized

*Marine Strategy Framework Directive

5.1. Monitoring of ecosystem effects

- Fish communities

To evaluate both the desired changes (stable or reduced but not collapsing cyprinid stocks and increasing stocks of piscivores) and undesired changes (invasions, increase of other planktivore fish) it is important to monitor the fish communities targeted by the fishery. To monitor changes in the species composition and abundances of coastal fish communities, we suggest using coastal Nordic multi-mesh gill nets annually in the summer (Thoresson 1993; Ljunghager & Karlsson 2020). Nordic multi-mesh gill nets are the standard survey nets used to monitor coastal fish communities along the Swedish and other Baltic Sea countries' coasts (HELCOM 2020). The gear is similar to the standard used in lakes (CEN 2015), but lacks the three smallest mesh sizes. Multi-mesh gillnets are well suited to follow up effects on the targeted fish community following biomanipulation (Olin 2005).

Time-series covering several years of annual monitoring can reveal population status and elucidate long-term trends in coastal fish communities (Olsson et al. 2012). Monitoring data should contain biological information such as individual size, age, and sexual maturity of targeted cyprinid species and piscivores (e.g. perch) to forecast future production in stocks (Östman et al. 2016), by using the indicators suggested by Sundblad et al. (2020).

Standardized fish monitoring with coastal Nordic multi-mesh gill nets are, however, relatively expensive, in the order of €10 000-15 000 per sampled site, and a reduced effort or collaboration with fishers will be required to reduce the costs. Currently, the NGO 'Race For The Baltic' have a contract with cyprinid fishers to report all catches in cyprinid gears. This is a cost-efficient method to follow abundance of some larger species (e.g. perch, pike), but not for fish smaller than commercial sizes, that are not caught but might replace the targeted cyprinid species. Thus, this fishery report system needs to be complemented with regular fishery independent monitoring. Yearly monitoring is desired but given the costs less frequent, i.e. every secondto thirdyear, can be motivated. With less frequent monitoring, it would be more difficult to separate changes in abundance or size from interannual sampling variation.

- Zooplankton abundances

Identification and enumeration of zooplankton (for example filter-feeding Cladocera and copepod species) is conducted with filtered water samples under a microscope (Horppila & Kaisalo 1990; Gorokhova et al. 2016). The abundance of zooplankton indicates the grazing pressure on phytoplankton (Lampert 1988). However, relatively large volumes of water are needed for good precision and accuracy of zooplankton abundance and composition, and the work is labour intensive. Unless a more mechanistic understanding of the processes and interactions in the system is needed, we do not see zooplankton abundance as a prioritized area for monitoring ecosystem effects of cyprinid fisheries.

- Zoobenthos

The diet of cyprinids is to a large extent constituted by zoobenthos. If cyprinids are reduced, this could result in a reduced predation pressure on zoobenthos. Zoobenthos are monitored in some lakes and there are guidelines for monitoring zoobenthos in the Baltic Sea (Evans & Leonardsson 2016). However, adequate sampling of zoobenthos requires several samples from the same area and the work is labour intensive and requires sorting of zoobenthos from sediment samples. Thus, we do not see monitoring of zoobenthos as prioritized in the monitoring of the effects of targeted cyprinid fishery.

- Phytoplankton abundance and chlorophyll α concentration

A major contributing factor to low water transparency is high phytoplankton and cyanobacterial biomass (Breukelaar et al. 1994; Cronberg 1999; Triest et al. 2016). Phytoplankton and cyanobacteria biomass and identity can be estimated from filtered water samples that are analysed under a microscope. But this is labour intensive and expensive. A proxy used for phytoplankton and cyanobacterial biomass is chlorophyll α concentration, which is used in most studies of biomanipulations. Chlorophyll α can be measured in situ with a device or a water sample can be analysed in a lab. An *in-situ* device is a cheaper option, but tends to have lower precision and usually requires rather high concentrations for accurate measurements.

As reduction of chlorophyll α is a major aim of biomanipulation, we think it is important to monitor the chlorophyll α concentration following fisheries targeting cyprinids. Again, this should preferable be measured with high frequency using optical sensors (eventual automated loggers if possible) repeated several times during spring and summer to capture seasonal dynamics in chlorophyll α.

- Submerged macrophytes

Submerged macrophytes have an important ecological function as substrate for fish reproduction and nutrient sedimentation. Percentage cover macrophyte species sensitive to turbidity, like *Chara* spp, *Ruppia* spp, and *Zostera* spp, might be key indicators for changes in water transparency following a biomanipulation targeting cyprinids (Blindow 2019; Eriksson et al. 2004). Submerged macrophytes are generally negatively affected by fast growing epiphytic filamentous algae that can thrive in eutrophic environment (Eriksson et al. 2009; Duffy et al. 2013). It would therefore be useful to estimate coverage of filamentous algae and submerged macrophytes. We therefore find it informative to monitor the distribution and abundance of macrophytes and filamentous algae in shallow and wave-sheltered areas. This can be relative expensive if done by snorkelling but aerial (drones) or satellite data may be possible to use to detect major changes in macrophyte distribution and abundance (Husson et al. 2014).

- Water transparency

Improvement in water transparency is a primary aim for biomanipulation targeting cyprinids. Water transparency can easily be measured as Secchi-depth with a white plastic disc with a diameter of 25 cm (Ådjers et al. 2006), or as the light attenuation coefficient with a LI-COR quantum sensor (Horppila & Kaisalo 1990). Although improvements in water transparency are usually observed relatively soon after cyprinid biomanipulations in lakes, the effect is likely less immediate and obvious in coastal areas because of generally higher water transparency (compared to shallow eutrophic lakes), high water exchange, and impact of waves and currents that can affect the water clarity (Wilas et al. 2016). We nevertheless suggest that water transparency is measured weekly to monthly spring to fall in areas targeted by cyprinid fisheries, since it is an easy and cheap measure and provides information about the eutrophication status and the extent of bioturbation (Bergström et al. 2016; Östman et al. 2017). It is important that water transparency is monitored repeatedly or continuously with automated data-loggers over the season to capture its high seasonal variation due to weather factors to scientifically assess changes in water transparency in a fishing area.

- Nutrient concentrations

External loading and internal recirculation of phosphorus and nitrogen are the ultimate causes for eutrophication and associated symptoms of poor water quality. Nutrient levels can be measured directly to predict and evaluate the success of biomanipulations (Horppila & Kaisalo 1990). Biomanipulations of cyprinids have generally low, but variable, impact on phosphorus and nitrogen levels in the water in lake systems (Bernes et al. 2015), unless the biomanipulation is complemented with chemical treatments (Al, Fe) that binds the available nutrients to the sediment. As the majority of the water masses in coastal areas are exchanged during a year, it seems unlikely to find local effects on nutrient levels as a result of cyprinid fisheries. We find it wiser to calculate how much nutrients that are removed by the fishery using data on the catch to estimate nutrient load (P: 0.75%, N: 2.5%; Mäkinen 2008) rather than monitoring nutrient concentrations directly from the water.

- Oxygen levels

Excess amounts of nutrients can result in high oxygen consumption when phytoplankton decompose. This in turn can result in hypoxic conditions or in the worst case, anoxic conditions in deeper water layers. Hypoxia can occur along the Baltic Sea coastal zone in sheltered highly eutrophic bays. If fisheries targeting cyprinids are focused on areas with historic risks of hypoxia, we find it important to monitor oxygen concentrations to assess if cyprinid fisheries may reduce hypoxic conditions. This is important at least during mid-summer when water temperatures are highest and there is a greater risk for anoxic conditions, but high frequency sampling or automated data-loggers are preferable.

5.2. Monitoring of contaminants in cyprinid fish from the Baltic Sea

We conclude that none of the examined substances were found to surpass the European maximum levels (MAL) in food, and hence, that cyprinids from the Baltic Sea can be sold and used for human consumption without any restrictions. Still, the Tolerable weekly intake (TWI) for some contaminants, primarily PFAS and dioxins and dioxin-like PCBs, was in the order of 100 g for a 60 kg person. Dioxin concentrations in Baltic Sea cyprinids were or on par with measured values of farmed (Norwegian) salmon (median 0.23 pg TEQ/g ww; Hannisdal et al. 2018). Thus, an increased consumption of cyprinids replacing farmed salmon as food would not affect intake of dioxins, but may increase intake of PFAS. On the other hand, if an increased consumption of cyprinids would add to existing consumption of seafood also dioxins and PCB intake may increase (depending on what is replaced). Regarding PFASs there is work in progress to decide MAL, so threshold values for what is safe to consume is likely to come. To be on the safe side of PFAS intake, a normal portion of cyprinid fish should not be consumed more than weekly.

The measured levels of contaminants in cyprinids in the Baltic Sea were similar to other wild fish species from Sweden (Karlsson & Viktor 2014, Waldetoft & Karlsson 2020). Muscle tissue from salmon and herring from the Baltic Sea have more than 10 times higher concentrations of dioxins and PCBs than cyprinids in the Baltic Sea (Cantillana & Aune 2012) so replacing these for cyprinids would lower at least dioxin and PCB intake.

If cyprinids are to become more used for human consumption, we recommend a more extensive screening (both in space and time) of especially PFAS, which showed considerable variation between species, locations or even samples of the same location, to better understand contamination levels and their variation in cyprinid fish. This is of extra importance if children will be consuming the fish on a regular basis in schools, since they weigh less than adults and may be more vulnerable to the toxic effects of several of these substances. Levels of contaminants differ between areas and there is no temporal variation in our data, so monitoring likely needs to be site specific, but would preferably also consider seasonal changed, especially including samples prior or during spawning when fish tend to be fattest (Aro et al. 2000; Szlinder-Richert et al. 2010).

5.3. Additional knowledge needs

Based on our literature review it is unclear if a targeted cyprinid fishery can reduce eutrophication problems and improve the status of the Baltic Sea.

Firstly, there may be positive effects of cyprinid fisheries that are not directly linked to the structure and function of the ecosystem, such as diversification of fisheries and increased domestic food production. However, as the ecosystem effects of cyprinid fisheries in Baltic Sea coastal areas are virtually unknown, ecosystem effects should be monitored. Initially, yearly seasonal monitoring of water transparency and monitoring of chlorophyll α concentrations in the areas where cyprinid fishing is already occurring would be valuable. In addition, it would be desirable to study macrophyte distributions and filamentous algae in an area prior to cyprinid fisheries and then repeat the study again 3-5 years after fishing has started. All monitoring of ecosystem variables in a fished area should be accompanied by measurements in nearby control areas where there is no cyprinid fishing to control for changes caused by other regional factors (temperature, salinity etc.).

Currently there are regional reference fish monitoring areas relatively close to sites where cyprinid fishing occurs today (Råneå in Bothnian Bay and Herrvik east of Gotland) where effects on fish communities may be detected. In addition, via cooperation with the organization Race For The Baltic, SLU has access to catch data, including bycatches from cyprinid fishing. Thus, changes in the targeted fish communities can be monitored using these sources of existing data.

The movement range of the targeted fish is a knowledge gap that would be possible to address in the Bothnian bay area where a targeted cyprinid fishery occurs at several sites. By marking and releasing undersized bream and ide, and preferably some of target size as well (with Floy-tags or T-bar), the movement and site fidelity of the fish could be studied to assess the spatial structure of stocks and estimate stock size in an area to establish reference and target levels for the fishery. In addition, population genetic studies can be informative of the spatial structure of stocks (Östman et al. 2017); however, genetic markers are not well developed for most cyprinid species.

For a more thorough scientific assessment of the impact of targeted cyprinid fisheries (longer perspective, 2-10 years), it would be desirable with more extensive evaluations in conjunction with a substantial cyprinid fishery to ensure that fishing actually affects cyprinid stocks.

6. Market incentives and limitations for increased targeted fishing for cyprinids in the Baltic Sea

If water quality is improved (e.g. higher transparency, less algae blooms) through a targeted cyprinid fishery, the recreational values of coastal areas will likely increase in several ways. The willingness to swim, bath and partake in other water are likely higher in clearer waters. Thereby tourism might increase, benefiting companies based around maritime or coastal activities. For example, the tourism industry on the Island Öland, has been adversely affected by algae blooms during the holiday season (Södergren 2014). Increased cyprinid fishing could also increase the income, and thereby the viability, of commercial coastal fisheries (Malmström & Waldo 2021).

Here, in cooperation with fishers and the NGO 'Race for the Baltic', we identify limiting factors and possibilities for cyprinid fisheries in the small-scale coastal fishery from a market and business perspective. It is important to stress that an increased cyprinid fishery should always be accompanied by assessments of stock status to avoid overfishing, see Sundblad et al. (2020) for examples.

The overall limiting factor for increasing human consumption of cyprinid fish in Sweden is currently the low awareness of cyprinid products that results in a low demand. Although an increased awareness of cyprinid products is a first step to increase the demand, Swedish seafood consumption is highly focused on the fillets of a few species, e.g. salmon and cod (Ziegler & Bergman 2017). An increased human consumption of cyprinids will therefore also require a change in attitude among consumers towards new species and fish products to increase the demand. Race For The Baltic and Stockholm County's 'ResursFisk' provide information and promote cyprinid fish products towards the public sector (municipalities), restaurants and the retail market.

In order for Swedish fishers to target cyprinids, this fishery has to be economically viable. This is only possible if there is a demand for these products. The single most important factor for an increased fishing targeting cyprinids is therefore likely an increased awareness and willingness to pay a "decent" price of cyprinid fish products. Currently fishers are being paid (in 2021) around €1-2 per kilo whole cyprinid fish. Still, cyprinid fish food products can be more expensive for the end consumer than similar products of minced fish from other fish species, e.g. salmon and cod. For fish that can be sold as fillets the remaining parts are basically free commodities and consumers only pay for the process and distribution of these processed products. As it is currently not possible to extract a fillet of cyprinid fish due to many bones, consumers have to pay the full price for food products based on the cyprinids. Some kind of obvious added value like “local fish” or “low/positive environmental impact” to the consumer will therefore likely be important when marketing cyprinid fish products relative other fish species.

Before highlighting cyprinid fish as sustainable human food, the sustainability of the fishery needs to be assessed. Consumers must be sure that cyprinids are not from overfished stocks, the fishery does not have negative ecosystem effects or on other fish species, the climate impact is low, and above all, that the fish is safe and healthy to consume.

Currently the supply of cyprinid food products seems to match the demand, but we think Swedish fishers can quite easily increase their supply. If demand increases the supply can likely increase. Cyprinid catches per unit effort tend to be large relative to many other coastal fish species, catches above 100 kg per catch are common. Therefore, the aggregated value of a catch can be substantial, despite a relatively low price per weight. Still, an economically efficient cyprinid fishery requires relative expensive equipment (pound nets), so there might be a need for fishers to make investments in order to increase catches. As net profit within small-scale fisheries in Sweden are generally small (Waldo & Lovén 2019), there are little financial resources available for new investments in expensive equipment, which can be seen as risky for the individual fisher.

Another issue for future cyprinid fisheries is that the type of fishing gear used may require exemptions from local fisheries regulations. In the Gulf of Bothnia, the risk of bycatch of salmon may constrain the use of pound nets, and in the southern parts of the Baltic Sea bycatch of eels might be the primary concern. SLU has in collaboration with “Race For The Baltic”, collected information on by-catches in commercial cyprinid fisheries. Preliminary results from Bothnian Bay and Gotland suggest that bycatches are around 10% or less relative to the targeted catch, and most bycatch can be released alive (Östman et al. unpublished results.). There might be local fishing regulations restricting gears for a targeted cyprinid fishery. In general, we do not see that bycatch (at least if released) or current fisheries regulations should restrict catches in cyprinid fisheries at a national level.

First-order buyers of cyprinids for human consumption seem to have the potential to handle and process considerably larger volumes of cyprinids compared to the current situation. Currently there are very few possibilities for selling cyprinids to first order buyers, besides selling on the private market. This results in logistic challenges in the distribution of the raw fish. This includes storage of fish in freezers and expensive transport relative to the value of the catch.

The processing of cyprinids for food today requires a minimum size of around 500 g. It corresponds to a frequently found size of 25 cm for bream and ide, but few roach reach 500 g (https://www.slu.se/institutioner/akvatiska-resurser/databaser/kul/). With machines that can process smaller fish, a larger part of the stock might be used, or the small fish might be used to process other products (e.g. fish sauce). There are biological reasons to avoid landings of smaller individuals, but from a business perspective it could be desirable. Catches from cyprinid biomanipulations in lakes or coastal areas are dominated by smaller individuals, which could also be used for human consumption. This would, however, also require an efficient distribution system.

We are confident that an increased demand and price for cyprinid fish would stimulate fishers to overcome problems with investment both in gears and distribution. However, changing consumer awareness and demand can be a slow and complicated process without any guarantee of success. There have been several examples of new dishes and food items that rapidly have become popular on the Swedish market, e.g. taco, sushi, chicken/turkey, plant-based meat and milk substitutes, and not least farmed Norwegian salmon. Beside salmon in the seafood sector, the consumption of "new" species like sea bream and tilapia have made it in to the market at a relatively low level (< 100 tons/year; Ziegler & Bergman 2017). What makes these products successful compared to others is not within our competence to analyse but changes in demand are clearly possible. In Sweden, the consumption of food from cyprinids is also increasing but will likely be the limiting factor for a national cyprinid fishery in the nearest future.

7. Concluding remarks

All measured contaminant levels were far below EU's Maximum Allowed Level and should therefore not face any problems with restrictions of selling and consumption, and can be considered safe to consume according the regulations of the Swedish Food Agency. Still, based on the Tolerable Weekly Intake, consuming cyprinids more than once a week should be avoided, since some samples had elevated concentrations of especially PFAS and the associated uncertainties about its toxicity. There was substantial spatial variation in contaminant levels between sampling sites. Thus, there is a need for additional site-specific monitoring of contaminants in cyprinids. Overall, however, it appears that cyprinid fish are much safer to consume compared to many other fish from the Baltic Sea.

Our review indicates that the ecosystem effects of increased cyprinid fisheries are uncertain and likely site dependent, but water transparency and eventually macrophyte distributions might increase. We believe that targeted cyprinid fisheries in the Baltic Sea will have a marginal impact on nutrient levels in the water column, as the effect of cyprinid reductions are uncertain and there is a large exchange of water masses between coastal and open sea areas. Still, cyprinid fishing can be a cost-effective method for local removal of nutrients in the Baltic Sea in the short-term time perspective.

As very little is known about ecosystem effects of targeted cyprinid fisheries in the Baltic Sea, more knowledge is needed. We suggest monitoring of Secchi depth, fish communities, macrophytes, chlorophyll α levels and eventually oxygen levels in areas with targeted cyprinid fisheries to evaluate any potential ecosystem effects. More large-scale controlled studies are required for scientific evaluation of ecosystem effects of cyprinid fisheries.

Although the interest for cyprinids as human food is increasing among fishers and the general public in Sweden, the consumer awareness and demand for the products is low. The need for fishers to invest in expensive gear and a relatively expensive distribution system, limit fishers' willingness to test and develop cyprinid fisheries.

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Appendix 1: Compilation of lake restoration studies in northern Europe used for analyses of this report.

Study lakes are divided into following biomanipulation categories: (A) Total fish reduction (rotenone or empty a lake) , (B) Fishing targeting cyprinids, (C) Fishing targeting cyprinids together with stocking of piscivores or (D) Fish targeting cyprinids together chemical/physical treatments, and (E) Stocking of piscivores without any direct reduction of fish stocks. Basic information on the study lakes and sources of the data; chemical and/or physical treatments of the lake (addition of Al, Ca or Fe, hypolimnetic aeration (O_2), dredging (dre) or flushing (flu)); cyanobacteria present (p) and dominant fish species before the manipulation (see Table 2 for codes for fish species); piscivores stocked during the manipulation; methods of fish removal (rotenone(ro), seine fishing (se),electrofishing (el), drainage (dr), fish kill (fk), trawling (tr), fish trap (ft), gillnetting (gn); amount of fish removed during the manipulation period; bycatch; catch usage (biogas (bg), animal feed (af), relocation (rl), human consumption (hc), fishmeal (fm), compost (co), export (ex), fish oil (fo), destruction (de)); start year and duration of the manipulation; fish stock reduction by the manipulation; mean values of total phosphorous, total nitrogen, and chlorophyll α (before/during/after) or (before/after) the manipulation period; Secchi depth, macrophyte and Daphnia responses, and condition of remaining fish individuals and/or the whole community after manipulation (increased (+), decreased (-), no change (+/-)); the outcome (successful (S), partially successful (PS) in some parameters, unsuccessful (US)) and durability of manipulation effects.

Table A1.1: Total fish reductions

Lake name[Reference]	Country	Surface area (ha)	Mean depth (m)	Dominant fish species	Method	Fish removal (kg ha^{-1})	Start (duration)	Reduction on fish stock (%)	TP (mg/l)	TN (mg/l)	Chl. α (µg/l)	Secchi depth	Macro.	*Daphnia*	Fish cond	Outcome (durability) (years)
Asklundvatn[1, 2]	NO	3.1	2	Cac	ro	790	1984 (1)	100%				(+)	(-)	(+)		PS (>4)
Cockshoot Broad[3, 4]	UK	3.3	1	Abr, Rr	se, el	396	1989 (2)	95%	(0.08/0.075/0.13)		(41/20/26)	(+)	(+)	(+)		S (>5)
Great Linford (Main Lake)[5]	UK	17	1.5	Abr, Rr	dr, se	396	1987 (2)	100%				(+)	(+)			S (>5)
Great Linford (St. Peters Lake)[5]	UK	2	1	Abr, Rr, Tt	dr, se	356	1987 (2)	100%					(+)			S (>5)
Haugatjern[6]	NO	9.1	7.6	Cl, Pf	ro	165	1980 (1)	100%	(0.045/0.033)			(+)	(+)	(+)		S (>3)
Helgetjern[6]	NO	12	1.8	Rr, Pf	ro	100	1984 (1)	100%	(0.18/0.085)		(110/55)	(+)		(+/-)		S (1)
Ijzeren Man[7, 8]	NL	52	1.2	Abr, Cyc	dr	709	1989 (1)	100%	(0.4/0/0.3)	(2.0/0/1.0)	(100/0/10)	(+)	(+)	(+)		S (3)
Lilla Stockelidsvatten[9]	SE	1	3	Rr	ro		1973 (1)	100%	(0.014/0.011/0.07)	(0.7/0.7/0.45)		(+)				PS (>4)
Lingese[10]	DE	38.8	6.7	Rr, Abr	dr	216	1995 (1)	100%	(0.14/0.05/0.04)		(137/25/6)	(+)		(+)		S (>8)
Llandrindod Wells[11]	UK	6.8	1.3	Sc, Cyc, Tt, Abr, Rr	dr, se	822	1991 (1)	100%	(0.24/0.20/0.04)	(0.15/0.03/0.14)	(58/74/37)	(+)				S (>5)
Maltanski[12]	PL	64	3.1	Abr, Rr, Pf	dr	398	1992 (4)	100%	(0.1/0.2)	(3.0/4.0)	(10/20)	(-)	(+/-)	(+)		U (1)

Maltanski[13]	PL	64	3.1	Abr, Rr, Cac	dr	319	1996 (1)	83%								U (0)
Mosvatn[6, 14, 15]	NO	46	2.1	Cl	ro	100	1987 (1)	100%	(0.05/0.04/0.03)	(0.68/0.32)	(23/23/7)	(+)	(+)	(+)		S (>1)
Ringsjön Östra[16-18]	SE	2050	6.1	Rr, Abr, Pf	fk, tr	243	1988 (1)	80%	(0.2/0.07/0.1)		(48/28)	(+)	(+)	(-)		S (>5)
Wirbel[19]	PL	11	1.8	Rr	ro	230	1990 (1)	100%				(+)		(+)	(+)	U (0)
Zwemlust[1, 20]	NL	1.5	1.5	Rr, Abr, Se	dr	1000	1986 (5)	100%	(1.2/1.0/0.8)	(2.2/1.5/1.5)	(250/12/20)	(+)	(+)	(+)		S (>5)

Table A1.2 Fishing targeting cyprinids

Lake name[Reference]	Country	Surface area (ha)	Mean depth (m)	Chem. Phys. treatment	Cyan.	Dominant fish species	Piscivores stocked	Method	Fish removal (kg ha^{-1})	Bycatch (kg ha^{-1})	Catch usage	Start (duration)	Reduction fish stock (%)	TP (mg/l)	TN (mg/l)	Chl. α (µg/l)	Secchi depth	Macro.	*Daphnia*	Fish cond
TARGETED FISH REDUCTIONS																				
Bleiswijkse Zoom[21]	NL	14.4	1.1			Abr, Cyc			670			1981 (2)					(+/-)	(+/-)		
Bälingesjön[22]	SE	43	2.8			Abr, Rr		se	103	5		2018 (2)		(0.03/0.02)	(0.52/0.46)	(18/9)	(+)			(+)
Duiningermeer[23]	NL	28	1		p	Rr, Abr		tr, se, ft, el	119		fm	1992 (1)	18%	(0.11/0.14/0.10)	(2.5/4.0/2.0)	(60/20/10)	(+)	(+)	(+/-)	
Dystrup Sø[24]	DK	26	1			Tt, El, Pf		gn, ft	230			1995 (6)		(0.21/0.33)	(2.5/3.5)	(55/142)	(-)			
Ejstrup Sø[24, 25]	DK	42	1.4			Abr, Rr		tr, gn, se	638	2%		1995 (4)	96% cypr.	(0.15/0.1/0.1)	(1.7/1.2/0.9)	(90/60/60)	(+)			(+)
Etujärvi[26]	FI	16	3.2		p	Rr, Abr, Pf		se, ft	348			1997 (4)	>75%	(0.03/0.04/0.04)	(0.65/0.7/0.6)		(+/-)		(+)	
Frisian lakes[1, 27]	NL	10000	2		p	Abr, Gc, Oe, Rr		tr, se, gn	215		hc	1985 (5)	27% cypr. yr^{-1}			(120/100/70)	(+)			(-)
Gridsted Engsø[28]	DK	30.4	1.6			Rr, Abr, Gc			160			2003 (3)	59%	(0.06/0.03)	(1.35/2.1)	(56/13)				
Hale Sø[24, 25]	DK	10	0.8			Rr, Pf		se	150			1996 (1)	75%	(0.18/1.1/2.0)	(1.8/3.9/1.96)	(25/22/67)	(+/-)		(+)	(+)
Hiidenvesi (Basin 1)[26]	FI	160	0.9		p	Rr, Lm		se, ft	44			1997 (4)		(0.09/0.08/0.09)	(1.0/1.1/1.3)		(+)		(+)	
Hiidenvesi (Basin 2)[26]	FI	260	2		p	Rr, Lm		se, ft	411			1997 (4)	>75%	(0.14/0.09/0.11)	(1.6/1.0/1.35)		(+)		(-)	
Hiidenvesi (Basin 3)[26]	FI	360	2.6		p	Oe, Rr		se, ft	153			1997 (4)		(0.05/0.06/0.05)	(0.8/1.2/1.0)				(+/-)	
Hiidenvesi (Basin 4)[26]	FI	970	11.2		p	Oe, Ala		se, ft	121			1997 (4)		(0.03/0.04/0.04)	(0.7/1.0/1.1)				(+/-)	
Häckebergasjön[29]	SE	76	2			Rr, Abr		gn, se	385	16	af, bg	2017 (4)		(+/-)			(+)	(+)	(+)	
Jyväsjärvi[30]	FI	337	7.2			Pf, Rr, Abr		ft, gn	294			2004 (3)	33% cypr.			(13/10/12)	(+/-)			(+)
Lehijärvi[26]	FI	704	6		p	Rr, Pf		se, ft	90			1997 (4)		(0.03/0.035/0.03)	(0.58/0.55/0.65)				(+)	
Loldrup Sø[28]	DK	39	1.2			Rr, Abr, Gc			618			1996 (5)	77%	(0.16/0.14)	(3.2/2.0)	(138/101)				
Nannewiid[7, 31]	NL	100	1			Rr, Abr			157			1993 (2)	84%	(0.45/0.39/0.17)	(5.5/5.5/2.5)	(170/150/70)	(+)	(+)	(+)	
Noorddiep[20, 32]	NL	4.5	1.6			Abr, Cyc			561			1987 (1)	71%	(0.2/0.4/0.5)	(2.2/1.9/1.55)	(55/10/12)	(+)	(+)	(+/-)	(+)
Nydam[24]	DK	2	1			Cac, Tt, Rr, Pf		se, ft, el	383			2004 (3)	75% cypr.	(0.23/0.12/0.08)			(+)			(+/-)
Nørresø[25]	DK	114	1.2			Rr, Abr			114	32%		1995 (2)		(0.08/0.05/0.04)		(46/19/9)	(+)			
Otalampi[26]	FI	31	3.3			Rr		se, ft	119			1998 (3)		(0.03/0.03/0.03)	(0.65/0.5/0.5)				(+/-)	
Pohjalampi[33]	Fi	61	3.2		p	Rr, Pf, Abr		gn	200			1993 (4)	80%	(0.027/0.023/0.015)		(20/12/8)				(+)
Ringsjön Västra[16, 17, 34]	SE	1484	3.1		p	Rr, Abr, Pf		tr, gn	150			1992 (1)	85%	(0.056/0.067/0.1)		(37/26)	(+)	(+/-)	(+/-)	
Ringsjön (Västra, Östra)[34, 35]	SE	3530	4.5		p	Rr, Abr, Pf, Sl		tr, gn	291		af	2005 (12)	85%	(0.098/0.07/0.05)			(+)	(+)	(+)	(+)
Rusutjärvi[26]	FI	133	2		p	Rr, Abr		se, ft	201			1997 (3)	>75%	(0.045/0.06/0.05)	(1.1/1.0/1.1)		(+/-)		(+)	
Rørbæk Sø[24, 25]	DK	84	4.3		p	Rr, Abr, Pf		tr, gn, ft	892	35%		1994 (12)	90% cypr.	(0.09/0.06/0.05)	(1.6/1.2/0.8)	(72/50/36)	(+)	(+)	(+)	(+)
Stubbergård Sø[28]	DK	150							851			2005 (4)		(0.24/0.26)	(2.7/1.5)	(142/83)				
Sätoftasjön[16-18]	SE	420	3		p	Rr, Abr, Pf		tr, se, gn, ft	800		af, bg	1989 (3)	60%	(0.09/0.08/0.04)		(58/35)	(+)	(+)	(-)	
Søbygård[36]	DK	40	1		p	Abr, Rr, Se		se, el, ft	100			1988 (1)	17%	(+/-)		(940/140)	(+)		(+)	(+)
Sövdesjön[37]	SE	272	3.4		p	Abr, Bb, Rr, Ala, Tt		se	385		bg, af	2017 (3)		(0.06/0.13)	(1.6/0.9)	(79/51)	(+)			(+)
Takajärvi[26]	FI	15	2.1			Rr, Abr		se, ft	295			1997 (4)	>75%	(0.043/0.037/0.03)	(0.7/0.7/0.7)		(+/-)		(-)	
Terra Nova[38]	NL	85	1.4		p	Rr, Abr, Tt		se, el, gn	262			2003 (3)	80%	(0.07/0.079)			(+)			
Trummen[39]	SE	76	1.1			Abr, Bb		se, gn	117		bg	2016 (1)		(0.043/0.02/0.016)		(40/20/10)	(+)	(+)	(+/-)	(+)
Vallentunasjön[40, 41]	SE	578	2.4		p	Abr, Rr, Ala		tr, gn, se	277		bg	2010 (6)		(0.05/0.07/0.06)	(1.3/1.4/1.4)	(30/35/35)	(+/-)			
Veluwemeer[1, 27]	NL	3400	1.5		p	Abr, Rr, El, Pf		se	75		hc	1993 (2)	80%	(0.53/0.04)		(70/60/10)	(+)	(+)		
Vesijärvi (Paimela Bay)[1, 42]	Fi	390	6.8		p	Oe, Abr, Ala		se, ft	445		hc	1992 (4)		(-)	(-)		(+)	(+)		
Vesijärvi (Komonselkä)[43, 44]	FI	1250	6.8		p	Abr		se, ft	66		co, ex	2002 (4)				(25/10/13)				
Vesijärvi (Laitalanselkä)[43, 44]	FI	2150	6.8		p	Abr		se, ft	79		co, ex	2002 (4)				(25/10/13)				
Vesijärvi (Kajaanselkä)[43 - 46]	FI	4400	6.8		p	Abr		se, ft	37		co, ex	2002 (4)		(0.03/0.015)		(25/10/13)				(+)
Væng Sø[24, 25]	DK	15.7	1.2		p	Abr, Rr		ft, el, gn	255			1986 (3)	50%	(0.15/0.12/0.10)	(1.0/0.6/0.8)	(78/35/50)	(+)	(+)	(+)	
Årungen[47, 48]	NO	120	8.1		p	Rr, Pf, El, Ana		gn	19 (El)			2004 (3)	88% cypr.	(0.13/0.05/0.03)			(+)	(+/-)		(+)
Äimäjärvi (Basin 1)[26]	FI	370	2		p	Rr		se, ft	257			1997 (4)	>75%	(0.07/0.075/0.07)	(0.95/1.0/0.95)	(+/-)	(+/-)		(+)	
Äimäjärvi (Basin 2)[26]	FI	480	3		p	Rr		se, ft	226			1997 (4)	>75%	(0.04/0.055/0.05)	(0.7/0.65/0.7)		(+/-)		(+)	
Ülemiste[49]	ES	975	3.4			Abr, Rr		se	160			2004 (3)		(0.048/0.042/0.036)	(1.5/1.2/1.2)	(30/22/21)	(+/-)		(+)	

Table A1.3 Cyprinid reductions and stocking of predatory fish

Lake name[Reference]	Country	Surface area (ha)	Mean depth (m)	Chem. Phys. treatment	Cyan.	Dominant fish species	Piscivores stocked	Method	Fish removal (kg ha^{-1})	Bycatch (kg ha^{-1})	Catch usage	Start (duration)	Reduction fish stock (%)	TP (mg/l)	TN (mg/l)	Chl. α (µg/l)	Secchi depth	Macro.	*Daphnia*	Fish cond
TARGETED FISH REDUCTION AND STOCKING OF PISCIVORES																				
Arreskov Sø[24, 25]	DK	317	1.9		p	Rr, Abr, Pf	El	se	230	48-73%		1992 (5)	90% cypr.	(0.24/0.11/0.19)	(2.9/1.5/2.4)	(144/39/48)	(+)	(+)	(+)	(+)
Bastrup Sø[24, 25]	DK	33	3.5		p	Abr, Rr	El	se, gn	213	11%		1995 (3)	39%	(0.08/0.06/0.07)	(0.97/0.87/0.69)	(35/21/22)	(+)			(+)
Bleiswijkse Zoom (Galgje)[20, 21, 50]	NL	3.1	1.1			Cyc, Bb, Sl	Sl	se, el	645			1987 (1)	85%	(0.4/0.17/0.2)	(3.0/1.7/2.1)	(100/15/50)	(+)	(+)	(+)	(+)
Borbjerg Møllesø[24, 25]	DK	13	1.3		p	Rr, Abr	Pf, El	se, el	1290	6%		1993 (8)	78%	(0.27/0.13/0.18)	(2.08/1.53/1.66)	(194/75/72)	(+)		(+)	(+)
Borup Sø[24, 25]	DK	9.5	1.1		p	Abr, Rr	Pf, El	gn, se	1056	1%		1996 (8)	74%	(0.24/0.17/0.10)		(120/50/20)	(+)		(+/-)	
Breukeleveen[1, 51]	NL	180	1.5		p	Abr, Rr, Pf	El	se, tr	93		hc (Sl)	1989 (1)	18%	(0.1/0.1/0.1)	(+/-)	(120/91/91)	(+/-)	(+/-)	(+/-)	(+)
Dalby Sø[25]	DK	15.2	1.4		p	Abr, Rr	El	gn, ft, se, el	307	22%		1995 (3)	59% cypr.	(0.27/0.19/0.24)		(43/115/115)	(+)		(+)	(+)
Dallund Sø[25]	DK	15	1.9			Abr, Rr	El	gn, se	226	19-44%		1995 (3)	48%	(0.13/0.09)			(+)			
Engelsholm Sø[24, 25]	DK	44	2.6		p	Rr, Abr	Pf	gn, se	766	7-40%		1992 (5)	61%	(0.14/0.15/0.08)	(2.5/2.0/0.9)	(94/94/40)	(+)			(+)
Feldberger Haussee[1, 3, 52]	DE	130	6.3		p	Rr, Abr	El, Pf	se	736			1985 (12)	25%	(1.1/0.5/0.1)		(25/17/15)	(+)	(+)	(+)	(+)
Fredriksborg Slotssø[25]	DK	22.3	3.5			Rr, Abr, Pf	Pf	gn, ft, se, el	623			1986 (3)	59%			(110/75/125)	(+)			(+)
Haderslev Dam[25]	DK	269	2		p	Rr, Abr	El	gn, tr, se	922	15%	fo	1992 (6)	35%	(0.34/0.3/0.3)		(220/170/150)	(+)		(+)	(+)
Hvidkilde Sø[28]	DK	64	2				El		190			1997 (2)		(0.2/0.18)	(1.3/1.1)	(94/39)	(+)			
Klein Vogelenzang[53]	NL	18	1.5		p	Abr, Pf	El		247			1989 (1)	50% cypr.	(0.28/0.4/0.22)	(3.0/4.4/2.5)	(122/147/60)	(+)	(+/-)	(+)	
Klejtrup Sø[24, 25]	DK	134	1.8		p	Rr	El	se, el	378			1994 (9)		(0.22/0.18/0.13)	(2.2/2.4/1.8)	(74/102/70)	(+)		(+/-)	(+)
Klokkerholm Møllesø[24]	DK	7.5	1			Rr, Cg	Stf	se, el	400			2017 (1)		(0.21/0.10/0.18)	(2.4/2.2/2.3)	(142/130/172)	(+)			(+)
Köyliönjärvi[54]	FI	1250	3		p	Rr, Abr, Oe	Sl, El	se	414		af	1992 (7)	81%	(0.09/0.1)		(80/80)				(-)
Kymijärvi[43]	FI	647	2.6		p	Abr, Rr	Sl, Cl, Oe, St	se, ft	468			1995 (17)	70% cypr.				(+)			(+)
Lading Sø[28, 55]	DK	47	1			Rr, Abr, Gc, Se	El		713			1998 (4)	75%	(0.15/0.15)	(2.1/1.3)	(59/50)	(+)			(+)
Maikkalanselkä fjärdarna[43]	FI	360	2.8		p	Pf, Abr, Bb, Rr, Aba	Sl, Cl	se	900			1999 (7)		(0.06/0.05/0.05)	(+/-)	(25/22/18)	(+/-)			(+)
Maribo Søndersø[24, 25]	DK	852	1.7		p	Abr, Rr	El	gn, tr, ft	155	26%	de, hc	1991 (2)	59%	(0.11/0.09/0.06)	(2.5/1.4/1.5)	(90/50/30)	(+)	(+)		(+)
Ramten Sø[24, 25]	DK	29	1.2		_	Rr, Tt, Gc	El	se, ft, el	590			1995 (7)	75%	(0.25/0.07/0.06)	(4.1/1.7)	(120/50/35)	(+)			(+)
Rugård Nørresø[24]	DK	36	1.8		p	Abr, Pf	El	se	258			2004 (2)	90% cypr.	(0.1/0.2/0.4)	(2.0/1.5/1.3)	(150/31/30)	(+)			(+)
Skærsø[24, 25]	DK	16	1.4			Rr, Abr	El	gn, se, el	281			1993 (5)	30%	(0.08/0.07/0.08)	(1.4/1.2/1.2)	(34/29/39)	(+/-)			(+)
Stubbe Sø[24]	DK	376	2.9		p	Rr, Abr, Pf	El	gn, tr, el	348			2000 (4)	72% cypr.	(0.10/0.07/0.08)	(1.85/1.3/1.4)	(60/39/60)	(+)			
Søbo Sø[24, 25]	DK	21	3.6		p	Abr, Rr	El	gn, se	395	28-45%		1994 (4)	49%	(0.08/0.07/0.07)	(2.0/1.7/1.8)	(50/40/60)	(+)		(+)	
Søerne[24]	DK	46.7	3		p	Rr, Abr, Ld	El	gn, se	696			2000 (4)	85%	(0.17/0.7/0.3)	(1.4/1.0/0.7)	(64/20/10)	(+)	(+)		(+)
Sövdeborgssjön[50, 56]	SE	11	2.5			Rr, Cl, Cac, Pf	Sl	tr, ro		1%		1980 (5)	20%	(0.062/0.071)		(28/25/28)	(+/-)		(+)	
Tillerup Sø[28, 55]	DK	6	2.8			Rr, Cyc	El	se, el	700			1996 (4)	85%	(0.13/0.11)	(1.6/1.5)	(38/27)	(+)			
Tueholm[28]	DK	15.4	1.2				El		800			1997 (5)		(0.2/0.1)						
Vallum Sø[28, 55]	DK	18	1.1			Rr, Abr, Gc, Se	El	se, ft, el	345			2000 (2)	70%	(0.14/0.09)	(2.3/1.2)	(93/67)	(+)			(+)
Vesijärvi (Enonselkä)[43, 45-46, 50, 57]	FI	2600	6.8		p	Rr, Oe	El, St, Oe	tr	392		co	1989 (4)	80% cypr.	(0.045/0.035/0.03)	(-)	(30/20/15)	(+)	(+)	(+/-)	(+)
Viborg Nørresø[25]	DK	123	6.1				El		417			1987 (5)		(0.1/0.12)	(2.1/1.9)	(51/60)	(+)			
Viborg Søndersø[25]	DK	146	3.8				El		1356			1987 (19)		(0.2/0.22)	(2.0/1.5)	(70/60)	(+)			

Table A1.4 Cyprinid reductions and chemical or physical treatments

Lake name[Reference]	Country	Surface area (ha)	Mean depth (m)	Chem. Phys. treatment	Cyan.	Dominant fish species	Piscivores stocked	Method	Fish removal (kg ha^{-1})	Bycatch (kg ha^{-1})	Catch usage	Start (duration)	Reduction fish stock (%)	TP (mg/l)	TN (mg/l)	Chl. α (µg/l)	Secchi depth	Macro.	*Daphnia*	Fish cond
TARGETED FISH REDUCTION AND CHEMICAL/PHYSICAL TREATMENTS																				
Bergundasjön (Södra)[39]	SE	432	1.9	Al		Abr, Bb		se, gn	405		bg	2015 (3)		(0.18/0.15/0.2)		(75/50/75)	(+)	(+)		(+)
Elsterstausee[58]	DE	50	2	flu		Cyc, Hm, Rr	El, Sl, Sg	dr	2860			1991 (3)	90% cypr.				(+)		(+)	
Enäjärvi[26, 43]	FI	492	3.4	O_2	p	Rr, Abr		se, ft	373			1993 (5)	46% cypr.	(0.08/0.12/0.08)	(0.8/1.1/0.9)	(80/60/40)	(+)			(+)
Finjasjön[18]	SE	1000	3	dre	p	Abr, Rr		tr	450		af	1992 (2)	88%	(0.21/0.05)		(60/20)	(+)	(+)	(+)	(+)
Fredriksborg Slotssø[24]	DK	22.3	3.5	Al	p	Rr, Abr		se	364	15-40%		2005 (2)	49%	(0.33/0.07)			(+)			(+)
Furesøen[24, 59]	DK	940	13.5	O_2	p	Abr, Oe		gn, se	226	28%		2003 (4)	15%	(0.20/0.10)	(-)	(-)	(+)	(+)	(-)	(+)
Hollands Ankeveense Plas[31]	NL	92	1.3	dre					255			1989 (4)	60%				(+)	(+)	(+)	
Kollelev Mose[24]	DK	5	1.5	Al, Fe, O_2	p	Rr, Cac	Pf		940			1998 (3)		(0.12/0.03/0.05)			(+)			
Pusulanjärvi[26]	FI	207	4.5	O_2	p	Rr, Abr		se, ft	182			1997 (4)		(0.045/0.055/0.05)	(0.7/0.9/0.9)				(+/-)	
Tiefenwarensee[60]	DE	141	9.6	Al, Ca, O_2		Hm, Rr, Abr, Bb	El, Ca	gn	48			2002 (3)	78%	(0.1/0.04/0.02)		(12/12/6)	(+)		(+)	
Torup Sø[28]	DK	20.2	3.9	O_2			El		100			1998 (3)		(0.08/0.05)	(1.8/1.1)	(44/23)				
Tuusulanjärvi[26]	FI	592	3.2	O_2	p	Abr, Rr			472			1997 (4)	>75%	(0.11/0.09/0.09)	(1.4/1.1/1.2)		(+)		(+)	
Växjösjön[39]	SE	79	1.9	Al, dre		Abr, Bb		se, gn	869		bg, hc	2016 (2)		(0.033/0.024/0.022)		(32/21/10)	(+)	(+)		(+)
Wolderwijd[1, 61]	NL	2650	1.5	flu	p	Abr, Rr	El	tr, se, ft	275		rl	1990 (3)	63%	(0.2/0.12/0.07)	(2.5/2.0/1.5)	(75/55/25)	(+)	(+)	(+)	

Table A1.5 Stocking of piscivores

Lake name[Reference]	Country	Surface area (ha)	Mean depth (m)	Chem. Phys. treatment	Cyan.	Dominant fish species	Piscivores stocked	Method	Fish removal (kg ha^{-1})	Bycatch (kg ha^{-1})	Catch usage	Start (duration)	Reduction fish stock (%)	TP (mg/l)	TN (mg/l)	Chl. α (µg/l)	Secchi depth	Macro.	*Daphnia*	Fish cond
STOCKING OF PISCIVORES																				
Bautzen[1, 2, 50, 62]	DE	533	7.4		p	Pf	Sl, El, Ana, Sg		0			1977 (14)		(0.17/0.45)		(+)	(-)		(+)	
Ferring sjö[25]	DK	320	1.5		p	Ga	Om		0			1992 (12)								
Gjersjøen[6, 50]	NO	270	23		p	Rr, Abr	Sl		0			1981 (6)	15%	(0.02/0.017)		(-)				
Glumsø Sø[28]	DK	23	1.3	Al			El, Oe		0			1997 (7)				(154/142/180)	(-)			
Lyng Sø[25]	DK	10	2.4		p	Rr, Abr	El		0			1990 (4)		(0.08/0.06/0.04)		(95/70/30)	(+)		(-)	(+)
Mutek[63]	PL	10.7				Rr, Abr	St, El, Pf, Sg, Om		0			1995 (5)	33%	(0.10/0.13/0.13)	(2.0/2.2/2.2)	(150/120/100)	(+)		(+)	(+)
Oldenor[25]	DK	37	1.8			Ga, Rr	Pf, El		0			1995 (3)	80% cypr.	(0.2/0.17/0.2)		(84/90/35)	(+)		(+)	(+)
Udbyover Sø[55]	DK	18	1.1			Rr	El, Pf		0			1994 (7)		(0.47/0.15)	(5.3/1.5)	(397/60)	(+)			(+)
Wirbel[19]	PL	11	1.8		p	Rr, Bb	El		0			1987 (5)								(+)

Table A1.6. *Codes for fishes used in Tables A1.1-A1.4.*

Code	Scientific name	Family	English name	Swedish name
Aba	*Abramis ballerus*	Cyprinidae	blue bream	faren
Abr	*Abramis brama*	Cyprinidae	bream	braxen
Ala	*Alburnus alburnus*	Cyprinidae	common bleak	löja
Ana	*Anguilla anguilla*	Anguillidae	eel	Europeisk ål
Bb	*Blicca bjoerkna*	Cyprinidae	white bream	björkna
Ca	*Coregonus albula*	Salmonidae	vendace	siklöja
Cac	*Carassius carassius*	Cyprinidae	crucian carp	ruda
Cg	*Carassius gibelio*	Cyprinidae	prussian carp	silverruda
Ch	*Clupea harengus*	Clupeidae	Atlantic herring	strömming
Ci	*Ctenopharyngodon idella*	Cyprinidae	grass carp	gräskarp
Cl	*Coregonus maraena*	Salmonidae	whitefish	sik
Cyc	*Cyprinus carpio*	Cyprinidae	common carp	karp
El	*Esox lucius*	Esocidae	Northern pike	gädda
Ga	*Gasterosteus aculeatus*	Gasterosteidae	three-spined stickleback	storspigg
Gc	*Gymnocephalus cernuus*	Percidae	ruffe	gärs
Hm	*Hypophthalmichthys molitrix*	Cyprinidae	silver carp	silverkarp
Ld	*Leucaspius delineatus*	Cyprinidae	sun bleak	groplöja
Li	*Leuciscus idus*	Cyprinidae	ide	id
Lm	*Lepomis macrochirus*	Centrarchidae	bluegill	blågälad solabborre
Oe	*Osmerus eperlanus*	Osmeridae	European smelt	nors
Om	*Oncorhynchus mykiss*	Salmonidae	rainbow trout	regnbåge
Pf	*Perca fluviatilis*	Percidae	yellow perch	Abborre
Pp	*Pseudorasbora parva*	Cyprinidae	stone moroko	Bandslätting
Rr	*Rutilus rutilus*	Cyprinidae	roach	Mört
Sa	*Salvelinus alpinus*	Salmonidae	Arctic char	Fjällröding
Sc	*Squalius cephalus*	Cyprinidae	chub	Färna
Se	*Rutilus erythrophthalmus*	Cyprinidae	rudd	Sarv
Sf	*Salvelius fontinalis*	Salmonidae	brook trout	Bäckröding
Sg	*Silurus glanis*	Siluridae	European catfish, wels	Mal
Sl	*Sander lucioperca*	Percidae	pike-perch, zander	Gös
Ss	*Sprattus sprattus*	Clupeidae	European sprat	skarpsill
St	*Salmo trutta*	Salmonidae	brown trout	öring
Stf	*Salmo trutta fario*	Salmonidae	river trout	*bäcköring*
Tt	*Tinca tinca*	Cyprinidae	tench	sutare

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Appendix 2. Fish sample preparation

Amount of fish tissue from each fish (ID) used for each sample. An empty cell means nothing from that fish was used for a sample. Total sample indicate the weight of aggregated samples .

ID	Area	Species	Size (cm)	Sample	Hg, Cd, P(g)	Tot. Hg, Cd, P (g)	PFAS (g)	Total PFAS (g)	Diox-in (g)	Total dioxin (g)	Fat (g)	Total fat (g)	PBDE (g)	Total PBDE (g)
1	Husum	Ide	43	RID 1	1.00		0.51		4.00		1.51			
2	Husum	Ide	42	RID 1	1.00		0.51		3.99		1.51			
3	Husum	Ide	41	RID 1	1.00		0.51		3.99		1.49			
4	Husum	Ide	43	RID 1	1.00		0.51		4.02		1.50			
5	Husum	Ide	45	RID 1	1.00		0.50		4.00		1.51			
6	Husum	Ide	43	RID 1	1.01		0.50		4.01		1.51			
7	Husum	Ide	42	RID 1	1.00	7.01	0.50	3.54	4.01	28.02	1.49	10.52		
8	Husum	Ide	45	RID 2	1.00		0.51		4.00		1.51			
9	Husum	Ide	44	RID 2	1.00		0.51		4.02		1.52			
10	Husum	Ide	40	RID 2	1.00		0.50		4.00		1.49			
11	Husum	Ide	45	RID 2	1.00		0.51		4.03		1.50			
12	Husum	Ide	47	RID 2	1.00		0.49		3.99		1.51			
13	Husum	Ide	44	RID 2	1.00		0.51		4.02		1.52			
14	Husum	Ide	39	RID 2	1.00	7.00	0.50	3.53	4.02	28.08	1.52	10.57		
15	Kalix	Ide	41	GID 1	2.00		1.10		9.99		3.50		0.98	
16	Kalix	Ide	43	GID 1	2.00		1.11		10.04		3.51		1.02	
17	Kalix	Ide	41	GID 1	2.00	6.00	1.11	3.32	10.02	30.05	3.51	10.52	1.04	
18	Kalix	Ide	46	GID 2	2.00		1.11		10.02		3.52		1.02	
19	Kalix	Ide	44	GID 2	2.00		1.09		9.99		3.49		1.00	
20	Kalix	Ide	44	GID2	2.00	6.00	1.10	3.30	10.01	30.02	3.52	10.53	1.02	6.08
21	Kalix	Bream	32	GBR	2.51		1.73		X		8.38			
22	Kalix	Bream	28	GBR	2.51	5.02	1.70	3.43	X	X	3.17	11.55		
23	Kalix	Roach	30	GM 1	0.50		0.30		5.01		1.01			
24	Kalix	Roach	31	GM 1	0.50		0.30		5.01		0.99			
25	Kalix	Roach	31	GM 1	0.50		0.32		5.00		0.99			

26	Kalix	Roach	29	GM 1	0.50		0.32		5.01		1.00	
27	Kalix	Roach	29	GM 1	0.51		0.32		5.02		1.02	
28	Kalix	Roach	30	GM 1	0.50		0.32		5.03		1.02	
29	Kalix	Roach	30	GM 1	0.50		0.31		4.99		1.01	
30	Kalix	Roach	30	GM 1	0.51		0.31		5.01		1.01	
31	Kalix	Roach	31	GM 1	0.50		0.29		5.01		1.00	
32	Kalix	Roach	29	GM 1	0.50		0.29		5.00		0.99	
33	Kalix	Roach	29	GM 1	0.50		0.32		4.99		1.02	
34	Kalix	Roach	28	GM 1	0.51		0.31		5.01		1.01	
35	Kalix	Roach	28	GM 1	0.51		0.32		5.00		1.01	
36	Kalix	Roach	29	GM 1	0.50	7.04	0.30	4.33	5.01	70.10	0.99	14.07
37	Kalix	Roach	29	GM 2	0.50		0.30		5.01		1.00	
38	Kalix	Roach	29	GM 2	0.50		0.30		4.99		0.98	
39	Kalix	Roach	21	GM 2	0.50		0.31		4.98		1.02	
40	Kalix	Roach	30	GM 2	0.51		0.31		4.99		1.00	
41	Kalix	Roach	30	GM 2	0.50		0.31		4.98		1.00	
42	Kalix	Roach	27	GM 2	0.50		0.31		5.02		1.02	
43	Kalix	Roach	25	GM 2	0.50		0.29		4.99		1.00	
44	Kalix	Roach	26	GM 2	0.50		0.31		4.84		1.03	
45	Kalix	Roach	25	GM 2	0.51		0.31		4.99		1.00	
46	Kalix	Roach	22	GM 2	0.50		0.30		5.00		0.99	
47	Kalix	Roach	25	GM 2	0.50		0.30		5.04		1.00	
48	Kalix	Roach	22	GM 2	0.50		0.32		4.98		0.99	
49	Kalix	Roach	23	GM 2	0.51		0.29		5.01		1.02	
50	Kalix	Roach	25	GM 2	0.50		0.29		4.99		1.00	
51	Kalix	Roach	21	GM 2	0.50	7.53	0.32	4.57	5.01	74.82	1.01	15.06
52	Kalix	Bream	36	GB 1	1.00		0.49		8.00		0.71	
53	Kalix	Bream	39	GB 1	1.01		0.51		8.01		0.70	
54	Kalix	Bream	43	GB 1	1.00		0.50		8.01		0.72	
55	Kalix	Bream	35	GB 1	1.00		0.51		8.00		0.71	
56	Kalix	Bream	48	GB 1	1.00		0.51		7.99		0.71	
57	Kalix	Bream	37	GB 1	1.00		0.51		7.99		0.71	
58	Kalix	Bream	31	GB 1	1.00		0.49		7.99		0.70	
59	Kalix	Bream	29	GB 1	1.00		0.50		8.01		0.71	
60	Kalix	Bream	35	GB 1	1.00	9.01	0.52	4.54	8.01	72.01	0.70	6.37
61	Kalix	Bream	32	GB 2	1.00		0.51		5.02		0.70	
62	Kalix	Bream	41	GB 2	1.01		0.50		5.03		0.71	
63	Kalix	Bream	32	GB 2	1.00		0.51		5.01		0.70	
64	Kalix	Bream	29	GB 2	1.00		0.50		5.02		0.71	
65	Kalix	Bream	26	GB 2	1.00		0.50		4.99		0.69	
66	Kalix	Bream	32	GB 2	1.01		0.51		5.02		0.69	
67	Kalix	Bream	46	GB 2	1.00		0.50		5.01		0.71	
68	Kalix	Bream	33	GB 2	1.01		0.51		5.01		0.70	
69	Kalix	Bream	45	GB 2	0.99		0.50		4.99		0.69	

70	Kalix	Bream	31	GB 2	1.00	10.02	0.49	5.03	5.01	50.11	0.71	7.01
71	Norrs-undet	Bream	55	NB 1	0.50		0.52		7.00		0.70	
72	Norr-sundet	Bream	29	NB 1	0.51		0.51		7.00		0.69	
73	Norr-sundet	Bream	30	NB 1	0.51		0.52		6.99		0.71	
74	Norr-sundet	Bream	49	NB 1	0.50		0.50		7.00		0.70	
75	Norr-sundet	Bream	47	NB 1	0.50		0.50		7.03		0.71	
76	Norr-sundet	Bream	40	NB 1	0.50		0.52		7.02		0.71	
77	Norr-sundet	Bream	47	NB 1	0.51		0.49		7.01		0.70	
78	Norr-sundet	Bream	39	NB 1	0.50		0.49		6.99		0.70	
79	Norr-sundet	Bream	37	NB 1	0.50		0.50		7.00		0.70	
80	Norr-sundet	Bream	32	NB 1	0.51		0.50		7.01		0.69	
81	Norr-sundet	Bream	32	NB 1	0.49		0.51		7.00		0.70	
82	Norr-sundet	Bream	33	NB 1	0.51	6.04	0.50	6.06	7.01	84.06	0.69	8.40
83	Norr-sundet	Bream	41	NB 2	0.50		0.51		4.00		0.70	
84	Norr-sundet	Bream	34	NB 2	0.50		0.50		3.99		0.71	
85	Norr-sundet	Bream	48	NB 2	0.49		0.52		4.02		0.70	
86	Norr-sundet	Bream	38	NB 2	0.51		0.50		4.02		0.71	
87	Norr-sundet	Bream	45	NB 2	0.50		0.51		3.99		0.70	
88	Norr-sundet	Bream	49	NB 2	0.50		0.50		4.01		0.69	
89	Norr-sundet	Bream	41	NB 2	0.51		0.50		4.00		0.70	
90	Norr-sundet	Bream	33	NB 2	0.51		0.49		4.02		0.71	
91	Norr-sundet	Bream	21	NB 2	0.51		0.51		3.98		0.71	
92	Norr-sundet	Bream	26	NB 2	0.51		0.51		4.01		0.69	
93	Norr-sundet	Bream	21	NB 2	0.51		0.50		4.00		0.70	
94	Norr-sundet	Bream	26	NB 2	0.50	6.05	0.49	6.04	3.98	48.02	0.69	8.41

95	Norr-sundet	Roach	25	NM 1	0.79		0.50		5.00		0.81			
96	Norr-sundet	Roach	24	NM 1	0.80		0.52		4.99		0.81			
97	Norr-sundet	Roach	22	NM 1	0.81		0.49		4.99		0.80			
98	Norr-sundet	Roach	31	NM 1	0.81		0.50		4.99		0.81			
99	Norr-sundet	Roach	30	NM 1	0.81		0.49		5.00		0.81			
100	Norr-sundet	Roach	23	NM 1	0.81		0.50		4.99		0.80			
101	Norr-sundet	Roach	25	NM 1	0.80	5.63	0.51	3.51	4.99	34.95	0.79	5.63		
102	Norr-sundet	Roach	20	NM 2	0.80		0.50		4.00		0.79			
103	Norr-sundet	Roach	21	NM 2	0.79		0.50		3.99		0.79			
104	Norr-sundet	Roach	25	NM 2	0.79		0.49		3.99		0.80			
105	Norr-sundet	Roach	20	NM 2	0.80		0.49		4.01		0.81			
106	Norr-sundet	Roach	24	NM 2	0.79		0.51		4.00		0.80			
107	Norr-sundet	Roach	24	NM 2	0.79		0.50		4.01		0.81			
108	Norr-sundet	Roach	26	NM 2	0.80	5.56	0.51	3.50	4.01	28.01	0.80	5.60		
109	Herrvik	Ide	47	HV 1	1.00		0.99		15.01		2.01		5.00	
110	Herrvik	Ide	51	HV 1	1.00		1.02		15.01		2.01		5.02	
111	Herrvik	Ide	48	HV 1	1.01		1.00		15.01		2.02		5.04	
112	Herrvik	Ide	48	HV 1	1.00		1.00		14.99		2.02		4.99	
113	Herrvik	Ide	48	HV 1	1.01	5.02	1.02	5.03	15.02	75.04	2.00	10.06	5.01	25.06
114	Herrvik	Ide	52	HV 2	1.00		1.01		15.00		2.00		5.01	
115	Herrvik	Ide	49	HV 2	0.99		1.01		15.01		2.00		4.98	
116	Herrvik	Ide	27	HV 2	0.99		1.00		14.99		2.01		4.98	
117	Herrvik	Ide	48	HV 2	1.02		1.01		15.02		2.00		5.01	
118	Herrvik	Ide	46	HV 2	1.00		0.99		15.01		2.00		5.00	
119	Herrvik	Ide	49	HV 2	1.00	5.00	1.00	6.02	14.99	90.02	1.99	12.00	4.99	29.97

120	Sör-sundet	Roach	20	SM 1	1.70		1.32		5.02		1.20	
121	Sör-sundet	Roach	21	SM 1	1.71		1.29		5.00		1.21	
122	Sör-sundet	Roach	24	SM 1	1.69	5.10	1.30	3.91	4.99	15.01	1.21	
123	Sör-sundet	Roach	25	SM 2	1.71		1.29		5.00		1.21	
124	Sör-sundet	Roach	23	SM 2	1.69		1.29		5.02		1.19	
125	Sör-sundet	Roach	25	SM 2	1.70	5.10	1.30	3.88	5.00	15.02	1.21	7.23

Appendix 3. Mercury, weight-normalized mercury, cadmium, phosphorous and fat

Measured concentration of mercury and cadmium and mass-normalised concentration of mercury in aggregated samples of cyprinid fish from the Baltic Sea. The table also show concentrations of phosphorus (P) and fat in the muscle tissue analysed.

Species	Site of capture	Average mass of individuals in aggregate sample	Cd	Hg	Hg mass-normalized	P	Fat
		kg	*mg/kg*	*mg/kg*	*mg/kg*	*mg/kg*	*g/100g*
Ide	Kalix	1.030	<0.001	0.0431	0.0418	2150	0.7
Ide	Kalix	1.138	<0.001	0.0422	0.0371	2210	0.99
Ide	Husum	1.067	<0.001	0.0423	0.0396	2290	1.2
Ide	Husum	1.188	<0.001	0.0435	0.0366	2300	2
Ide	Herrvik	1.625	<0.001	0.093	0.0572	2570	0.55
Ide	Herrvik	1.819	<0.0009	0.109	0.0599	2690	0.84
Bream	Kalix	0.589	<0.001	0.0609	0.1034	2480	0.54
Bream	Kalix	0.540	0.00117	0.0761	0.1408	2320	0.92
Bream	Kalix	0.326	<0.001	0.0427	0.1312	2480	0.58
Bream	Norrsundet	0.936	<0.001	0.0441	0.0471	2760	0.57
Bream	Norrsundet	0.734	<0.001	0.0397	0.0541	2090	0.58
Roach	Kalix	0.343	<0.001	0.0827	0.1334	2190	0.62
Roach	Kalix	0.228	<0.001	0.074	0.1471	2330	1
Roach	Norrsundet	0.224	<0.001	0.18	0.3607	2350	0.55
Roach	Norrsundet	0.157	<0.001	0.148	0.3515	2360	0.84
Roach	Sörsundet	0.134	<0.001	0.0986	0.2513	2540	2.4
Roach	Sörsundet	0.176	<0.001	0.0611	0.1376	2210	0.73

Appendix 4. Dioxins and dioxin-like PCBs

Measured concentration and lower and upper bound sums of concentrations of dioxins and dioxin-like PCB in aggregated samples of cyprinid fish from the Baltic Sea.

Species/Location	Ide						Bream				Roach					
Contaminent (ng/g)	Kalix		Husum		Herrvik		Kalix		Norrsundet		Kalix		Norrsundet		Sörsundet	
2,3,7,8-tetraCDD	<0.21	<0.093	<0.17	<0.11	<0.16	<0.12	<0.078	<0.11	0.3	0.18	<0.098	<0.14	<0.11	<0.2	<0.19	<0.16
1,2,3,7,8-pentaCDD	<0.32	<0.15	<0.33	<0.23	<0.21	<0.16	<0.2	<0.16	0.27	0.33	<0.15	<0.2	<0.12	<0.33	<0.27	<0.24
1,2,3,4,7,8-hexaCDD	<0.46	<0.51	<0.56	<0.62	<0.24	<0.3	<0.42	<0.4	<0.26	<0.25	<0.52	<0.37	<0.21	<0.34	<0.31	<0.32
1,2,3,6,7,8-hexaCDD	<0.46	<0.51	<0.56	<0.62	<0.24	<0.3	<0.42	<0.4	0.36	0.46	<0.52	<0.37	<0.21	<0.34	<0.31	<0.32
1,2,3,7,8,9-hexaCDD	<0.46	<0.51	<0.56	<0.62	<0.24	<0.3	<0.42	<0.4	<0.26	<0.25	<0.52	<0.37	<0.21	<0.34	<0.31	<0.32
1,2,3,4,6,7,8-heptaCDD	<0.41	<0.27	<0.48	<0.25	<0.47	<0.54	<2.1	<1.4	<1.3	<0.84	<0.31	<1.5	<0.4	<0.84	<0.51	<0.52
Oktaklordibenso-dioxin	<1.6	<0.51	<1.3	<1	<0.94	<0.69	<2.4	<1.6	<1.8	<1.3	<0.99	<1.8	<0.53	<1.8	<0.82	<0.69
2,3,7,8-tetraCDF	<0.12	<0.29	<0.36	<0.48	<0.54	<0.39	<0.12	<0.14	1.9	2.1	<0.28	<0.22	<0.84	<0.55	<0.14	<0.14
1,2,3,7,8-pentaCDF	<0.19	<0.22	<0.22	<0.16	<0.15	<0.2	<0.25	<0.22	0.72	0.26	<0.22	<0.22	<0.78	<0.37	<0.18	<0.21

2,3,4,7,8-pentaCDF	<0.19	<0.22	<0.22	<0.16	<0.15	<0.2	<0.25	<0.22	<0.24	<0.075	<0.22	<0.22	<0.78	<0.37	<0.18	<0.21
1,2,3,4,7,8-hexaCDF	<0.32	<0.34	<0.35	<0.46	<0.28	<0.21	<0.25	<0.26	<0.25	<0.22	<0.35	<0.22	<0.25	<0.2	<0.2	<0.39
1,2,3,6,7,8-hexaCDF	<0.32	<0.34	<0.35	<0.46	<0.28	<0.21	<0.25	<0.26	0.38	<0.22	<0.35	<0.22	<0.25	<0.2	<0.2	<0.39
1,2,3,7,8,9-hexaCDF	<0.32	<0.34	<0.35	<0.46	<0.28	<0.21	<0.25	<0.26	<0.25	<0.22	<0.35	<0.22	<0.25	<0.2	<0.2	<0.39
2,3,4,6,7,8-hexaCDF	<0.32	<0.34	<0.35	<0.46	<0.28	<0.21	<0.25	<0.26	<0.25	<0.22	<0.35	<0.22	<0.25	<0.2	<0.2	<0.39
1,2,3,4,6,7,8-heptaCDF	<0.3	<0.18	<0.32	<0.18	<0.47	<0.4	<1.4	<1.2	2.6	<0.59	<0.18	<1	<0.35	<0.89	<0.56	<0.72
1,2,3,4,7,8,9-heptaCDF	<0.3	<0.18	<0.32	<0.18	<0.47	<0.4	<1.4	<1.2	<1.4	<0.59	<0.18	<1	<0.35	<0.89	<0.56	<0.72
Oktaklordibenso-furan	<1.2	<0.39	<1	<0.79	<0.73	<0.53	<1.9	<1.2	<1.4	<0.99	<0.76	<1.4	<0.41	<1.4	<0.64	<0.53
sum WHO-PCDD/F-TEQ lowerbound	0	0	0	0	0	0	0	0	0.89	0.77	0	0	0	0	0	0
sum WHO-PCDD/F-TEQ upperbound	0.45	0.33	0.48	0.43	0.39	0.3	0.33	0.31	1	0.87	0.34	0.35	0.43	0.48	0.36	0.38
PCB 77	<4.7	<6.9	<12	<12	<18	<18	<3.3	<12	<13	<12	<8.1	<4.6	<11	<5.2	<4.3	<5.6
PCB 126	<3.2	<1.2	3.7	2.5	3.3	3.2	<3.5	<2.9	<3.9	<0.84	<2.3	<1.3	1.6	0.73	<1.5	<0.78
PCB 169	<5.3	<2.9	<8	<4.4	<0.5	<0.59	<2.7	<3.1	<4.4	<1.6	<5	<1.8	<0.94	<0.97	<0.69	<1.4
PCB 81	<8.8	<5.5	<3.6	<3.3	<0.94	<0.94	<5.2	<16	<15	<13	<9.1	<9.4	<1	<0.87	<0.82	<1.9
PCB 105	<39	50	86	90	88	120	<29	<26	65	50	63	<45	110	57	<34	<51
PCB 114	<11	<10	<8.1	<47	<3.7	<4	<15	<16	<8.8	<7	<25	<27	<3.7	<3.7	<4.6	<1.2
PCB 118	99	170	250	230	310	360	<100	<100	320	210	280	<93	470	220	<140	<160
PCB 123	<11	<6.4	<2.8	<12	5	8.4	<17	<17	10	<7.5	<10	<28	6.1	9	<4.5	<2.4
PCB 156	20	28	49	46	40	51	<18	<19	74	47	36	<32	120	71	<18	<26
PCB 157	<18	<10	8.9	<23	8.9	9.7	<24	<23	<15	<13	<14	<39	15	4.9	<6.6	<5.4

PCB 167	<19	<12	25	29	29	32	<20	<21	51	31	19	<33	71	46	<11	<12
PCB 189	<41	<32	<18	<61	3.1	<4.6	<22	<21	13	<11	<33	<37	9.7	11	<5.3	<1.6
sum WHO-PCB-TEQ lowerbound	0.0036	0.0076	0.38	0.26	0.34	0.34	0	0	0.016	0.01	0.012	0	0.19	0.086	0	0
sum WHO-PCB-TEQ upperbound	0.15	0.15	0.46	0.31	0.36	0.36	0.14	0.12	0.17	0.11	0.29	0.15	0.2	0.095	0.058	0.098
sum WHO-PCDD/F-PCB-TEQ lowerbound	0.0036	0.0076	0.38	0.26	0.34	0.34	0	0	0.906	0.78	0.012	0	0.19	0.086	0	0
sum WHO-PCDD/F-PCB-TEQupperbound	0.6	0.48	0.94	0.74	0.75	0.66	0.47	0.43	1.17	0.98	0.63	0.5	0.63	0.575	0.418	0.478

Appendix 5: Non-dioxin like PCB

Measured concentration (ng/g) and lower and upper bound sums of concentrations of non-dioxin like PCBs in aggregated samples of cyprinid fish from the Baltic Sea.

Sample	Species	Location	PCB 28	PCB 52	PCB 101	PCB 138	PCB 153	PCB 180	PCB6, sum "lowerbound"	PCB6, sum "upperbound"
			ng/g	*ng/g*	*ng/g*	*ng/g*	*ng/g*	*ng/g*	*ng/g*	*ng/g*
GID 1	Ide	Kalix	<0.056	<0.09	<0.18	<0.17	0.25	0.12	0.37	0.86
GID 2	Ide	Kalix	<0.066	<0.094	<0.18	0.27	0.47	0.17	0.91	1.2
RID 1	Ide	Husum	<0.085	<0.13	<0.21	0.45	0.81	0.28	1.5	2
RID 2	Ide	Husum	<0.091	<0.14	<0.19	0.43	0.82	0.26	1.5	1.9
HV 1	Ide	Herrvik	<0.05	<0.14	0.22	0.47	0.88	0.21	1.8	2
HV 2	Ide	Herrvik	<0.093	<0.15	0.28	0.58	0.76	0.21	1.8	2.1
GB 1	Bream	Kalix	<0.023	<0.071	<0.13	<0.21	<0.27	<0.1	0	0.8
GB 2	Bream	Kalix	<0.044	<0.065	<0.1	<0.21	<0.27	<0.1	0	0.79
NB 1	Bream	Norrsundet	<0.066	<0.19	0.62	0.9	1.7	0.78	4	4.3
NB 2	Bream	Norrsundet	<0.06	<0.16	0.36	0.56	0.95	0.48	2.3	2.6
GM 1	Roach	Kalix	<0.061	<0.091	<0.2	0.36	0.61	0.23	1.2	1.6
GM 2	Roach	Kalix	<0.021	<0.074	<0.13	<0.19	0.28	0.12	0.41	0.82
NM 1	Roach	Norrsundet	<0.037	<0.18	0.79	0.99	2.5	0.83	5.1	5.3
NM 2	Roach	Norrsundet	<0.052	<0.088	0.31	0.78	1.1	0.47	2.7	2.8
SM 1	Roach	Sörsundet	<0.038	<0.057	<0.13	<0.24	<0.38	<0.16	0	1
SM 2	Roach	Sörsundet	<0.032	<0.067	<0.18	<0.35	<0.46	<0.17	0	1.3

Appendix 6: PFAS and PFOS

Concentrations of detected PFAS-congeners in fish from the Baltic Sea. 'ND' are non-detectable concentrations (<Limit of Detection). Values within parentheses are in between Limit of Detection and Limit of Quantification (maximum level including errors) and should be interpreted with caution. Values in bold indicate measured concentrations above the detection limit. For PFDS, PFBA, PFHxA, PFHpA, PFDA, PFTriDA, FOSA were all samples below the detection limit.

Species	**Location**	**PFOS** *[ng/g]*	**PFOA** *[ng/g]*	**PFNA** *[ng/g]*	**PFHxS** *[ng/g]*	**∑PFOA +PFOS +PFNA +PFHxS** *[ng/g]*	**PFBS** *[ng/g]*	**PFPeA** *[ng/g]*	**PFUnDA** *[ng/g]*	**PFDoDA** *[ng/g]*
Ide	Husum	ND	ND	ND	ND	0	ND	ND	ND	ND
Ide	Husum	ND	ND	ND	ND	0	ND	ND	ND	ND
Ide	Kalix	ND	ND	ND	ND	0	ND	ND	ND	ND
Ide	Kalix	ND	ND	ND	ND	0	ND	ND	ND	ND
Ide	Herrvik	ND	ND	ND	ND	0	ND	(0.28)	ND	ND
Ide	Herrvik	**1.5**	ND	(0.19)	ND	***1.63***	**16**	**0.72**	**1.6**	**0.61**
Bream	Kalix	**3.6**	**1.7**	**2.0**	ND	**7.2**	ND	ND	(0.29)	ND
Bream	Kalix	ND	ND	ND	ND	0	ND	ND	ND	ND
Bream	Kalix	ND	ND	ND	ND	0	ND	ND	ND	ND
Bream	Norrsundet	ND	ND	ND	ND	0	ND	ND	ND	ND
Bream	Norrsundet	ND	ND	ND	ND	0	ND	(0.24)	(0.22)	ND
Roach	Kalix	ND	ND	ND	ND	0	ND	ND	ND	ND
Roach	Kalix	ND	ND	ND	ND	0	ND	ND	ND	ND
Roach	Norrsundet	ND	ND	(0.32)	ND	***0.32***	ND	**1.13**	**1.8**	**0.76**
Roach	Norrsundet	ND	ND	ND	ND	0	ND	ND	ND	ND
Roach	Sörsundet	ND	ND	ND	ND	0	ND	(0.37)	(0.44)	ND
Roach	Sörsundet	ND	ND	ND	ND	0	ND	ND	ND	ND
Limit of detection		*0.19*	*0.09*	*0.19*		*0.09*	*0.19*	*0.19*	*0.19*	
Limit of Quantification		*0.63*	*0.31*	*0.63*		*0.31*	*0.63*	*0.63*	*0.63*	

Appendix 7: PBDE

Measured concentration (µg/kg) and lower and upper bound sums of concentrations of non-dioxin like PCBs in aggregated samples of cyprinid fish from the Baltic Sea. Values in bold indicate measured concentrations above the detection limit.

Sample	GID 1	HV 1	HV 2
Species	Ide	Ide	Ide
Location	Kalix	Herrvik	Herrvik
BDE 28	<0.10	<0.050	<0.050
tetraBDE	<1.0	<0.50	<0.50
BDE 47	<0.10	**0.11**	**0.17**
pentaBDE	<0.50	<0.50	<0.50
BDE 99	<0.10	<0.050	<0.050
BDE 100	<0.10	<0.050	<0.050
hexaBDE	<1.0	<0.50	<0.50
BDE 153	<0.10	<0.050	<0.050
BDE 154	<0.10	<0.050	<0.050
heptaBDE	<2.0	<1.0	<1.0
oktaBDE	<2.0	<1.0	<1.0
nonaBDE	<10	<5.0	<5.0
dekaBDE	<10	<5.0	<5.0
dekabrombifenyl (DeBB)	<10	<5.0	<5.0
hexabromcyklododekan(HBCD)	<10	<5.0	**8.9**