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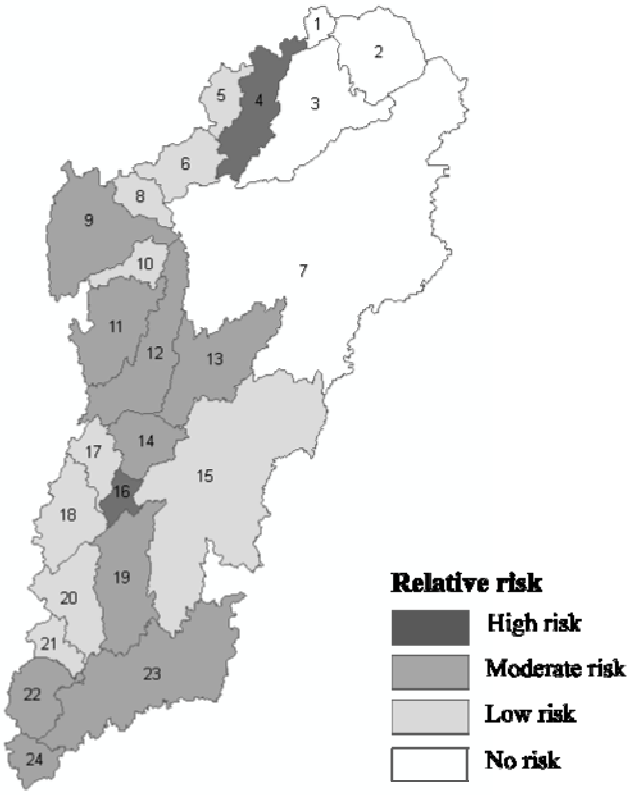

Ecological Risk Assessment of the River Basin of Göta Älv

– A study according to the PETAR procedure

Master of Science Thesis

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Division of Environmental Systems Analysis
CHALMERS UNIVERSITY OF TECHNOLOGY
Göteborg, Sweden, 2009
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Cover:
[Map showing the relative ecological risk in the different subareas within the river basin of Göta älv. More information about the relative risk can be found on pp. 20 and pp. 57.]

[Chalmers Reproservice]
Göteborg, Sweden 2009

In loving memory of Jan Rogbeck

I would like to dedicate this thesis in loving memory of my mentor and friend Jan Rogbeck who recently and suddenly passed away. Your support and thoughts have had a large impact on me; you were my source of inspiration. You are invaluable. I wish that your life would not have ended so soon.

Abstract

The ecological risks to the receptor organisms salmon and trout in the Göta älv river basin posed by five groups of pollutants; pharmaceuticals, nutrients, pesticides, metals and organic pollutants, have been studied in this thesis. The ecological risks were characterised with regard to both subareas within the river basin as well as with regard to the different pollutants. This study was performed according to the PETAR procedure, a procedure of ecological risk assessment, and the data used was obtained from both literature and model calculations.

Two subareas were identified as being of higher risk than the others; one area around Trollhättan and one located at Älvängen. In both of these areas, there is a relatively large wastewater treatment plant located that emits large amounts of several pollutants. Regarding the risk characterisation of the different pollutants, pharmaceuticals were identified as being of the highest risk to the aquatic ecosystem. For the pharmaceuticals, both literature and model calculations indicate that the environmental concentrations in water are higher than the corresponding PNEC values. For the pharmaceutical propranolol, the model calculations showed that the environmental concentrations were six times higher than the PNEC value. T herefore, the pharmaceuticals in Göta älv may cause adverse effects to the aquatic ecosystem. The results for the pesticides and organic pollutants are inconclusive; there are large differences in expected concentrations between the literature and the model calculations. One explanation for this may be differences in dilution between the sites studied in the literature and the conditions in Göta älv. Concentrations of metals and nutrients are low, close to background concentrations, and are therefore believed not to pose a risk to salmon and trout.

Due to the lack of data, both of site specific and general data for some of the pollutants, there are uncertainties within these results. The results from this study should therefore be seen as indicative and they can be used to prioritise the need for further studies. To confirm these results, site-specific measurements and/or more detailed model calculations are necessary. There is a need to perform more detailed risk assessments for pharmaceuticals, pesticides and organic pollutants since the knowledge about these pollutants is incomplete and regional studies like this one indicate that they may be problematic in the environment. Further studies can increase the understanding of the risks that they pose and how this risk is best managed.

Keywords: Ecological risk assessment, regional risk assessment, relative risk model, aquatic ecosystem, Göta älv, pharmaceuticals, pesticides, organic pollutants, metals, nutrients.

Acknowledgements

I would like to express my gratitude to all people that have helped me throughout this thesis. I would like to give a special thanks to my two supervisors Professor Sverker Molander and PhD Maria Paulsson who have supported and helped me throughout this entire study. Your input has been very valuable to me when deciding on the goal and scope as well as formulating the aim of this study.

In the process of this thesis, many people have helped me, especially during the gathering of data. I am very thankful for your help. Without your input, I would not have had enough data in order to perform the analysis.

Finally, I would also like to thank my family and friends who have been there for me, and who have listened to all of my thoughts and ideas.

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1 Introduction

Anthropogenic activities can give rise to many adverse effects; it can for example cause negative effects on the health of both humans and ecosystems. Many of the past anthropogenic activities have been proven to cause adverse effects. Among examples of such effects are for one the London smog, which occurred during the 1940s and the 1950s (Harrop, 2002). High concentrations of atmospheric pollutants resulted in the death of nearly 5000 people due to heart and respiratory failure. A second example is the use of DDT, which caused thinning of the eagle eggshells (Naturvårdsverket, 2008). A third example is the emissions of estrogens from wastewater treatment plants, which can feminize the reproductive organs in male fish (Naturvårdsverket, 2008b). To prevent adverse effects caused by present and future human activities, risk assessment can be used as a tool to predict possible adverse outcomes. The results of the assessment can then be communicated and used as a foundation for decisions regarding risk management.

The process of performing risk assessments were originally developed to protect the health of humans, which reflected the anthropocentric view of the environment. In that view, the humans were considered to be the entity worth protecting. During the 1970s, many adverse effects were noticed in the environment around the world, which gained a lot of attention and concern from the public. As a result, actions to prevent these adverse effects were initiated and the focus of risk assessment was changed so that it could also be applied to the environment. In Sweden, ecological risk assessments have previously by tradition mainly been focused on risks posed by emissions of metals and organic pollutants such as PCB and DDT. As a consequence, there is a wide knowledge base today about the effects caused by these pollutants, while knowledge regarding risks posed by other pollutants is much less. Risk assessments of contaminated sites in Sweden generally focus on “old” substances (e.g. heavy metals, petroleum hydrocarbons, PAH, PCB etc.) and not on “new” substances such as pharmaceuticals, which may be much more relevant in terms of risk. In order to make good predictions of risk, a holistic approach is necessary in risk assessment where all relevant pollutants are considered. By having a holistic view, the pollutants that pose the highest risk can be identified. The information about high-risk pollutants can then be used to design an efficient procedure for prioritising where further investigations are needed as well as prioritising remediation actions. Since the society only have limited funds for investigations and remediation, it is important that these funds are directed towards the areas where they are most needed. Another problem is that the spatial scope for prioritising sources of pollution is very narrow in Sweden, commonly with focus on small individual industrial areas, thereby missing to put the risk of the individual site into a regional perspective. Without having a regional perspective, it is difficult to tell what the risk of the individual site is compared to its surroundings, thereby making an efficient risk management process difficult. With the new Water Directive that has been adopted by the European parliament, new demands are put on ecological risk assessments. According to this directive, risk assessments performed should account for risks in an entire catchment area (Vattenmyndigheten, 2000). The new water directive and the need for and efficient method for prioritising high-risk pollutants indicate that there is a need for regional risk assessment. By using regional risk assessments, the most important sources of pollutants can be identified within a region, for example a specific drainage area where large amounts of pollutants may be emitted. This information can then guide further investigations and remediation, achieving an efficient risk management process.

Due to many reasons, the river basin of Göta älv constitutes an interesting catchment area to perform a risk assessment of. The Göta älv river basin has for a long period of time been exploited by humans and a wide variety of activities have been performed in the area, such as agriculture, commercial and industrial activities etc. As a result of these activities, the physical landscape has been altered, many places have been polluted and the water chemistry of the river has changed (Jacobsen and Johansson, 1999). At the same time, the river valley is hosting a wide variety of ecosystems with a high social and ecological value. As the situation is today, many areas within the river basin have been identified to possibly be in need of remediation due to contamination of metals and organic pollutants according to the Swedish procedures for ecological risk assessment, called MIFO (Naturvårdsverket, 1999). At the same time, both the industries and societies in the river basin emit substances that can cause negative effects on the ecosystem. Many of these substances belong to a group of pollutants where there is a lack of information regarding the risks that they pose. By performing a regional risk assessment for the entire river basin, it is possible to identify subareas within the river basin that are of highest risk. Also, by comparing the traditional stressors to the "new" stressors, it is possible to compare the risks that they pose and identify which are of highest risk of causing adverse effects. By knowing these facts, it is then possible to perform more detailed investigations within the high-risk areas to specify which factors that affect the risk and how it can be reduced. After that, an efficient risk management plan that deals with the areas and stressors that that pose the highest risk can be formulated.

1.1 Aim and goal

The general aim of this master thesis was to perform a regional risk assessment using the PETAR procedure for ecological risk assessment for the river basin of Göta älv, including a semi-quantitative assessment of the ecological risks present in the river. In more detail, this master thesis aims at answering the following questions:

- Is there a general risk for adverse ecological effects within the river basin of Göta älv?
- Can any of the assessed stressors be identified as being of particularly high risk to the ecosystems?
- Are there any subareas within the river basin that are of particularly high risk for adverse ecological effects?
- Is the PETAR procedure applicable to an area such as the river basin of Göta älv?

The goal of this thesis is to help improve the risk management process by providing a basis for decisions that can indicate where further investigations are needed to better characterise the risk within the river basin of Göta älv.

This ecological risk assessment was performed as a literature study, using available information. To visualize the results from the assessment, Geographical Information Systems (GIS) was to be used to construct maps showing the spatial distribution of stressor sources and sensitive areas such as habitats within the study area.

1.2 Study limitations

Limitations for the study have been made, which are listed below:

- This study only covers ecological effects, human health aspects have not been considered.
- No measurements have been performed during the study. All data have been obtained from available literature. It has therefore been necessary to make extrapolations and estimations to generate data that applies to the Göta älv river basin.
- The study area does not include the Göteborg harbour, since there are too many stressor sources to consider.
- Since there are a large number of stressors that are being emitted within the study area, it was not possible to evaluate them all. This study has therefore been limited to assessing a selection of chemical stressors.
- The study will only focus on the first two tiers of the PETAR procedure; the third tier that is the local risk assessment will not be performed.

2 Background

2.1 Ecological risk assessment

Risk assessment is a tool that is used within a large range of fields, which link science to the decision making process in order to help make decisions when the knowledge about the present and the future are uncertain. Examples of fields in which risk assessment is used as a tool are economics, medicine and ecology. Risk can be defined as the chance of an adverse effect with specific consequences to occur (Burgman, 2007). The aim of a risk assessment is thus to evaluate the likelihood that an adverse event with specific consequences would occur, as well as to evaluate the magnitude of these consequences in a systematic way. A central part in risk assessment is therefore to identify stressors that can cause adverse effects and receptors that are susceptible to the stressors as well as transportation and exposure pathways. In general, a stressor can be defined as the elements of a system that cause an unwanted outcome (Burgman, 2007). In ecological risk assessment, a stressor can be defined as an entity that can induce an adverse effect, where the entity can be of physical, biological or chemical origin (Moraes, 2002). The receptor is defined as the entity that is exposed to the stressor. It can for example be an ecosystem in an ecological risk assessment, a society in economical risk assessment or the human body in medical risk assessment. Since all risk assessments involve a mixture of uncertainties, it is also important to evaluate and communicate the nature and extent of these uncertainties in order to keep the assessment transparent and to determine the reliability and usefulness of the assessment. One of the uncertainties with risk assessment is connected to the definition of the concept ‘adverse effects’. The definition of an adverse effect is largely determined by people’s values and preferences, and it can therefore be considered as arbitrary. As a result of this arbitrariness, risk assessment has been criticised as a biased tool by the social sciences (Moraes, 2002). Despite that, risk assessment is still considered as a useful tool in decision making due to its transparent and systematic approach.

Ecological risk assessment (ERA) is one of the types of risk assessment that are commonly performed. ERA has been defined by the United States Environmental Protection Agency (U.S. EPA) as the process that evaluates the likelihood that adverse ecological effects may occur or are occurring as a result of exposure to one or more stressors (U.S. EPA, 1998). This process is based on a systematic approach of gathering, structuring, digesting and analyzing large amounts of data (Moraes, 2002). To guide the process of ERA, different procedures have been developed around the world. The U.S. EPA formulated one of the first ERA procedures in the document “A Framework for Ecological Risk Assessment”, which was published in 1992 (U.S. Environmental Protection Agency, 1992). This document was later revised and further developed and is now replaced by the “Guidelines for Ecological Risk Assessment" (U.S. Environmental Protection Agency, 1998). Another example of a procedure for ecological risk assessment is the so-called “Technical Guidance Document on Risk Assessment”, which was developed by the European Chemicals Bureau (2003). Today, there exist many different procedures, which to a great extent are based on the same central steps. These steps are some type of problem formulation, which consist of planning and scoping of the study, followed by an analysis step where the exposure and effects are characterised. The procedures usually end with a risk characterisation step, where the risk is estimated based on the data gathered in the analysis step.

The PETAR procedure is another ecological risk assessment method. It can be used to assess multiple kinds of stressors across a range of scales, at the same time as it accounts for

restrictions in data availability and acquisition (Moraes, 2002). The procedure was originally developed for use in developing countries where data availability and acquisition often is limited. The acronym stands for a Procedure for Ecological Tiered Assessment of Risks and the procedure is based on the Guidelines for Ecological Risk Assessment (U.S. EPA, 1998). As for many of the ERA procedures, the PETAR procedure is based on the same central steps mentioned above; i.e. a problem formulation step, an analysis step and a risk characterisation step. What is special for this procedure is that is aims at assessing the risks in entire regions instead of just assessing risks in small areas as most ERA procedures do. The PETAR procedure ranks the risks for different subareas within the study area with the aim to identify high-risk areas. The areas at high risk can then be given priority when deciding on remediation actions. A more detailed description of the PETAR procedure is given in the following section.

2.2 The PETAR procedure

The PETAR procedure is a three-tiered evaluation of risk where the method starts with large geographic areas and successively works towards smaller geographic areas with increased resolution of the results (Moraes, 2002). As seen in **Figure 1**, in which the main structure of the PETAR procedure is visualised, these three tiers are called Preliminary assessment, Regional risk assessment and Local risk assessment. After the PETAR procedure, the results are communicated to the risk manager, who then can make a decision regarding how the risk should be managed.

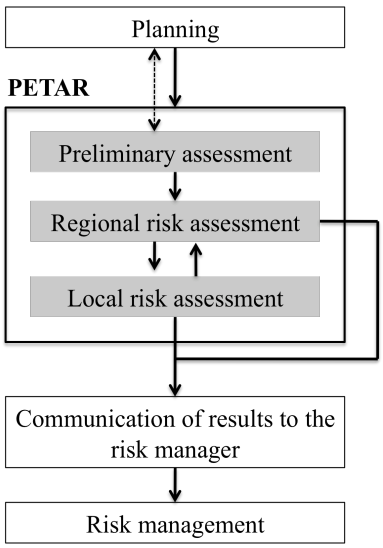


Figure 1 The overall process of the PETAR procedure
Source: Moraes, 2002

Prior to the PETAR procedure, it is necessary to have a planning phase where the goal and scope of the assessment are stated. To formulate the goal and scope of the assessment, a group containing a risk assessor, a risk manager and other interested parties identify possible regulatory needs as well as public concerns of interest to study. The result of this process, the scope, should include the scale of the study (geographical and temporal), the nature of the study, whether it should be retrospective or prospective, and finally it should also state what ecological value should be protected. In the following sections, the first two tiers of the PETAR procedure will be presented in more detail, since these two tiers are of focus in this study. The last tier, the Local risk assessment will only be described shortly.

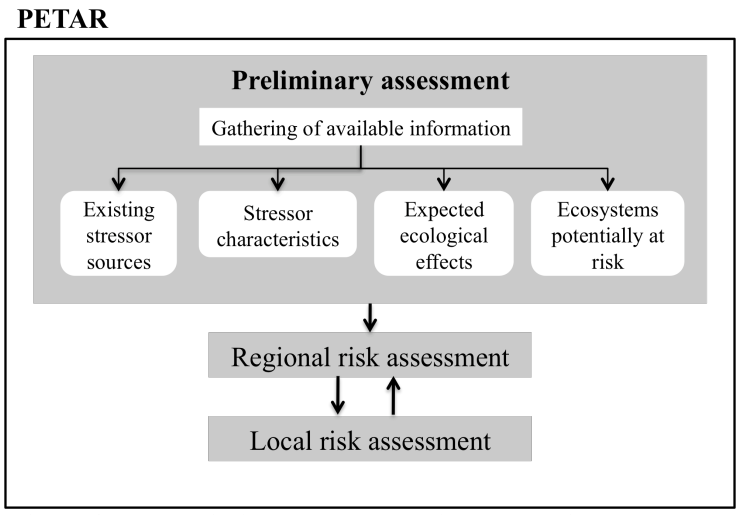


Figure 2 The preliminary assessment in the PETAR procedure
White rectangles represent steps in the process and rounded rectangles represent products of the assessment.
Source: Moraes, 2002

The step following the planning phase is the preliminary assessment. This part of the procedure is focused on gathering available information about the situation of the study area. The information that is gathered is then compiled and is used to guide the second tier, the regional risk assessment, in the procedure. Below in **Figure 2**, the steps of the preliminary assessment are presented. The process of gathering available information is represented by a white rectangle in the figure. The type of gathered information can be divided into four different groups; existing stressor sources, stressor characteristics, expected ecological effects and ecosystems potentially at risk. All of these four categories of information contribute to creating a picture of the current situation in the study area. For example, by knowing how many and what type of stressor sources that is located in the study area, it is possible to estimate which stressors will be emitted and in what amount. When knowing which stressors are emitted, one can list possible ecological effects that can occur as a result of the emissions. To know if the possible effects are likely or not, it is also necessary to know which types of ecosystems and species those are available in the study area, since there are species differences in toxic response. After having gathered this information, it is possible to focus the following tiers on the stressors that constitute the highest risk and the species that are most susceptible to the emitted stressors. The gathered and compiled information constitutes the results of the preliminary assessment, which are represented by rounded rectangles in the figure.

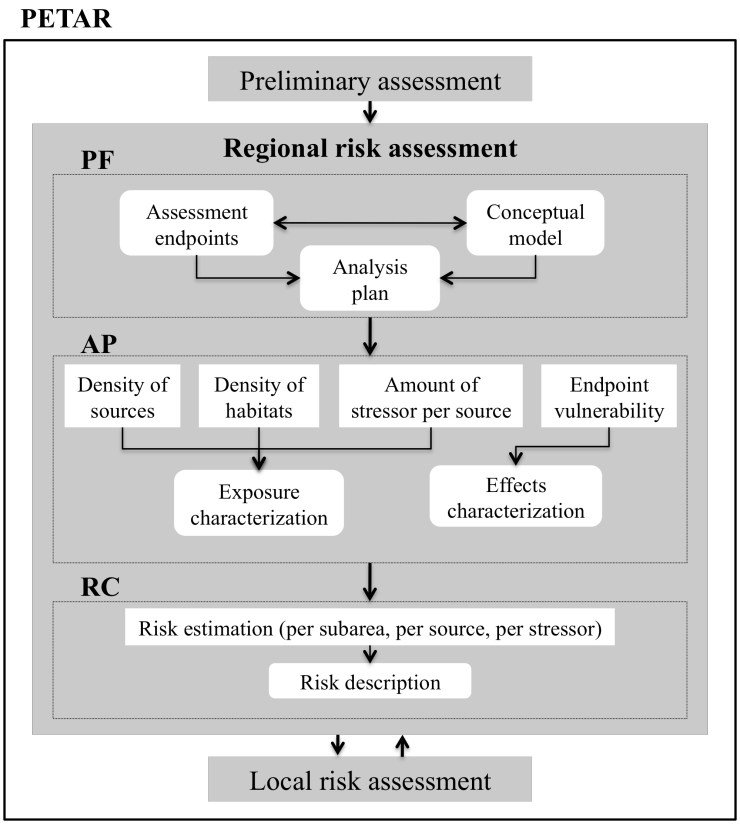


Figure 3 The regional risk assessment in the PETAR procedure
PF: problem formulation, AP: analysis phase, RC: risk characterisation.
White rectangles represent steps in the process and rounded rectangles represent products of the assessment.
Source: Moraes, 2002

The intermediate tier is called regional risk assessment and it is a semi-quantitative risk assessment that covers large geographical areas. This tier is based on the results from the preliminary assessment. The regional risk assessment is divided into three different parts, problem formulation, analysis phase and risk characterisation, which can be seen in **Figure 3** below. The problem formulation aims at describing the aim and purpose of the regional risk assessment by identifying appropriate assessment endpoints, making a conceptual model and formulating an analysis plan. The assessment endpoint and the conceptual models are to a large extent formulated based on the information that was obtained in the preliminary assessment. When these two have been formulated, it is possible to formulate the analysis plan, which states how the analysis step will be performed. The plan should state which method that will be used in order to assess the exposure and effects characterisations. After having finished the problem formulation phase, the analysis phase starts. In the analysis phase, data is gathered and compiled according to the problem formulation. In the PETAR procedure, the exposure is characterised by assessing the density of sources, density of habitats and amount of stressor emitted per source, while the effects are characterised by looking at endpoint vulnerability to the studied stressors. The regional risk assessment is then completed by the risk characterisation phase. In the risk characterisation, the risk is ranked with respect to different aspects. For example, the risk of different subareas within the entire studied region can be determined by using ranking methods. Another type of risk characterisation that can be performed is with regard to individual substances by calculating quotients between predicted environmental concentrations, PEC, and predicted no effect concentrations, PNEC. The aim of the regional risk assessment is to identify areas or stressors in the study region that is of high risk and in need of further investigation.

The last tier is the local risk assessment and it is a quantitative evaluation, which is also divided into three different parts in the same way as the regional risk assessment. The main difference between these two tiers is that during the local risk assessment, the object of study is a smaller geographical area and more detailed information is assessed.

2.3 Dealing with uncertainty in risk assessment

When assessing risk, it is important to analyse the uncertainties of the results in order to assess the reliability of the obtained results. Uncertainties can arise due to many different factors such as incomplete knowledge, disagreement between information sources, variability, model structure uncertainties etc (Morgan and Henrion, 1992). Uncertainty that arise from incomplete knowledge is called epistemic uncertainty. Within this group of uncertainties, there are two main sub-groups (Burgman, 2005). One is variability, which exists due to natural variations such as heterogeneity within a population. Collecting additional data cannot reduce this kind of uncertainty, although the variability can be better understood and therefore better estimated. The other type of uncertainty is called incertitude and it reflects the lack of knowledge about parameters and models. This type of uncertainty can be reduced by additional data. To minimize the uncertainty, it is important to understand uncertainties linked to both variability and incertitude.

When performing risk assessments, models are often used to describe the reality. Models are always simplifications of the reality. No matter how detailed, a model can never be complete. Models are therefore inherently uncertain. The simplifications in a model can arise due to many different factors; two common factors are lack of knowledge or too high complexity of the reality. These simplifications can sometimes be performed unwittingly, which makes them hard to identify.

Literature data and measurements are also used in risk assessment. These sources of information can also contain different degrees of uncertainty depending on how the information was obtained. When performing measurement campaigns, the uncertainties in the results can be reduced by using data from several samples taken at different locations and occasions to account for natural variability. A single sample is not very representative as it just represents a snapshot of the actual state. A good sampling strategy can therefore account for more of the natural variability. When analyzing the results, it is important to treat all samples in the same way to avoid measurement errors and systematic errors. The reliability of the results of a study is determined by the sampling and analysis strategies used. To be able to assess the reliability of data, it is therefore necessary to know the methodology.

In risk assessment, uncertainty analysis and sensitivity analysis is used to assess the adequacy and reliability of the data that is used in the study. The uncertainty analysis aims at identifying the source of the uncertainty and how it could affect the results. The sensitivity analysis instead, aims at quantifying the uncertainties by looking at how variations in the input could affect the output of a model. By performing these two analyses, the types and the quantity of uncertainty within the results of a risk assessment can be better understood and be implemented in the results, thereby making the results more informative for decision makers who are using them.

2.4 The Göta älv river basin

The river Göta älv is Sweden's largest river with regard to water flow, and its entire catchment area constitutes an area of approximately 50,000 km^2 or 10 percent of the total area of Sweden. As a part of this large catchment area is Lake Vänern, Sweden's largest lake. The river basin of Göta älv constitutes a much smaller area, and covers only the area where the tributaries drain directly to the river. The parts of the catchment area where the tributaries drain to Lake Vänern are not included in this study. Göta älv drains from Lake Vänern at the city of Vänersborg and flows in a southwest direction towards the west coast of Sweden. Before it reaches the west coast, at the city of Kungälv, the river is divided into two separate outlets called Göta älv and Nordre älv. The majority of the water flow discharges to the ocean via Nordre älv (Göta älvs vattenvårdsförbund, 2006; Länsstyrelsen Västra Götaland, 2002). Göta älv continues to flow south after the division and drains to the ocean at the city of Göteborg. The study area reaches from Vänersborg in the north to the junction of Göta älv and Säveån, just before the Göteborg harbour. The area that drains to Säveån is not included in the study area. In total, the length of the study area is approximately 90 km and it covers an area of 1,440 km^2 (SMHI, 2008).

There are lot of industries in the Göta älv river valley and approximately 170,000 people live in the communities in the region, not including the population of the municipality of Göteborg with approximately 500,000 inhabitants (Statistiska centralbyrån, 2008). These communities use the river as both their freshwater reservoir and as a receiving medium for their wastewater, which could create a conflict. Besides the industrial activities and the communities in the river valley, there is a lot of agriculture and farming since the soil in the river valley consists of fertile clay soil. The river valley also serves as a vital transportation corridor in the region, providing routes of transportation by ship, automobile and train. The current distribution of land use within the study is dominated by forests, out of which some is used for forestry. Arable land and open land that is often used for haysel and grazing in the agriculture constitutes the second and third largest parts of the area. The communities only make up for approximately a fifth of the total area.

The physical landscape in the catchment area of Göta älv is diverse and holds many different ecosystems, which have high social values. The landscape in the catchment can be divided into four main types of landscapes. In the northern part of the river valley, between Vänersborg and Lilla Edet, steep shore embankments, up to 20 m high, dominate the landscape together with gorges created by streams. On the hillsides facing the river, there are virgin coniferous forests as well as deciduous forests that have high species diversity. In this area, there are also areas kept open by grazing, which holds many species (Göta älvs vattenvårdsförbund, 2006). South of Lilla Edet, the landscape levels out and the river valley becomes wider. Here, bog meadows and areas with reed characterise the landscape in the river valley and a large part of the river valley is situated at surface level. Orchids can be found on many of the bog meadows in this area. In this part of the river valley, the land is used for pasture and haysel since the river brings a lot of nutrients to the meadows. The area is also an important location for many bird species, which use it as a resting place and/or breeding place. The third type of landscape consists of the estuaries that form at the outlet of Nordre älv and Göta älv where the freshwater from the river mixes with salt water from the ocean. Estuaries are known to be highly productive areas, due to the constant infusion of nutrients from the river, resulting in a rich fish and bird life. Even so, there is usually only a limited number of species that can survive in the estuary permanently due to the variations in salinity and water temperature (Nationalencyklopedien, 2009). The estuaries of Nordre and Göta älv are not included in this study. The river basin of Göta älv consists of a larger area

than just the river valley. The majority of the river basin is located on the eastern side of the river. This area constitutes the fourth and last type of landscape in the catchment area of Göta älv. The river valley is delimited on both sides by approximately 100 m high hillsides made of gneiss, diabase and granite (Göta älvs vattenvårdsförbund, 2006). Coniferous forests dominate the landscape of the tables surrounding the river valley.

Besides the terrestrial ecosystems in the river valley, the river also holds an aquatic ecosystem. In the Göta älv and its tributaries, there is a diversity of habitats that suits many different species. Most freshwater fishes live permanently in the river while the marine fishes visit the river occasionally. 37 freshwater species of the total 59 freshwater species living in Sweden have been found in the Göta älv at some occasion: This probably makes the Göta älv the river with the highest species diversity in Sweden with regard to freshwater fishes (Jacobsen and Johansson, 1999). Among these species, there are several that are listed as being endangered of extinction. The cause of this is believed to be the anthropogenic activities in the area that has changed the water chemistry of the river and created physical and biological barriers for the fish. Within the river basin, there are several natural reserves located.

3 Planning phase

3.1 Method

The study was initialised with the planning phase of the PETAR procedure. In this phase, the goal and scope of the study was formulated. First the study area was selected. This selection was based on the characteristics of the Göta älv river basin with all of the anthropogenic activities in the area and the current pollution problem in combination with the wide variety of highly valued ecosystems. Another contributing factor to the selection of study area was the fact that there are many ongoing or planned remediation actions within the study area.

After the study area had been selected, a screening study was performed in order to identify potential stressor sources. Based on the identified stressor sources, groups of stressors that were emitted from the sources were then identified. Among these groups, some were selected for further study here. Within each selected stressor group, a couple of substances were then selected to represent the group. To do so, a literature study was performed to gather data regarding the different stressor groups. The substances were selected according to the following criteria:

- The selected substances should be representative for the group
- The selected substances should be of environmental concern

Substances of environmental concern were defined as substances that fulfil any of the following criteria:

- Potentially toxic to the selected receptor organism
- Emitted in large quantities
- Persistent

The selection of what ecological value that should be protected, and accordingly what ecosystem/species that should be studied, was based on which ecosystems that was most likely to be exposed to the emitted stressors. To get an opinion of what was valued as important by stakeholders and risk managers, contact was taken with Lars Molander, fishing adviser at Länsstyrelsen and Per-Erik Jacobsen at Sportfiskarna. Also, in order to use available literature as a source of data for the study, the study was determined to be retrospective.

3.2 Results

The results from the screening study, that was performed in order to select relevant stressors to study, can be seen in the conceptual model shown in **Figure 4** below. The stressor sources in the model were identified based on past and present human activities in the study area. In the figure, different kinds of stressors that were relevant for the study area have been identified as well as their sources and transportation pathways.

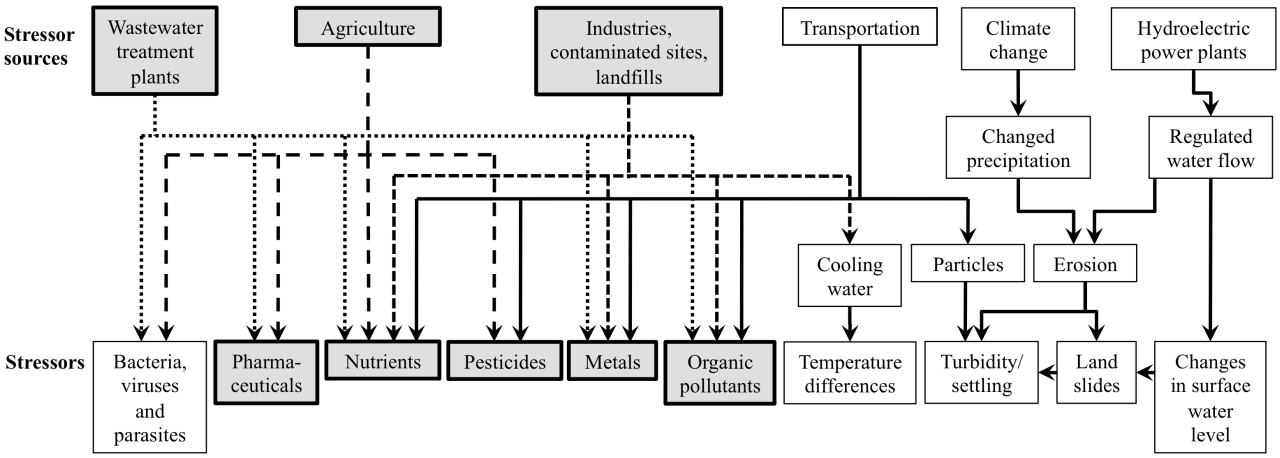


Figure 4 Conceptual model linking stressor sources to stressors
The stressors and stressor sources in grey boxes were selected for further study

The grey boxes represent the stressors and stressor sources that were chosen for further investigation in this study. The study was focused on chemical stressors and their respective stressor sources. The chemical stressors were selected since it would be interesting to compare the potential ecological effects caused by traditional chemical stressors such as metals and nutrients that have been studied for a long period of time in Göta älv to new chemical stressors such as pharmaceuticals, pesticides and organic pollutants. Even though transportation is an important source for many of the selected chemical stressors, it was not selected for further studies since the emissions from transportations are very diffuse and to a large extent emitted to the atmosphere. Because atmospheric transportation of stressors can be long-ranged, it was difficult to predict where the stressors would end up in the study area. The study was therefore focused on stressors deposited in the Göta älv river basin, while the main part of the stressors from transportation within the study area is likely to end up outside of the study area. Climate change is not considered as a stressor source in this study since this study is retrospective while a study of the effects of climate change needs to be prospective. The effects of hydroelectric power plants were not included in the study since they do not emit chemical stressors during operation.

The selected stressor sources in the study were wastewater treatment plants, agriculture, industries, landfills and contaminated sites. **Table 1** summarises the connection between the selected stressors and their respective stressor sources. In the table, as well as in **Figure 4** above, it can be seen that each stressor source emits several types of stressors.

Table 1 The selected stressor sources and selected groups of stressors that the emit

Wastewater treatment plants	Agriculture	Industries, contaminated sites and landfills
Pharmaceuticals	Pharmaceuticals	Nutrients
Nutrients	Nutrients	Metals
Metals	Pesticides	Organic pollutants (PAHs, HBCD etc.)
Organic pollutants (PAHs, HBCD etc.)		

For each group of stressors, a couple of substances that were assessed as being of environmental concern were selected as representatives for the different groups. The selection of metals was based on a ratio of measured environmental concentrations in Göta älv and literature LC_{50} (the concentration that is lethal for 50 percent of the individuals in a population) for rainbow trout. The substances that obtain the highest ratios (toxic unit) were assessed as the ones being most likely to cause toxic effects to the aquatic ecosystem. This

ratio was calculated for metals that were known to be toxic, see in **Table 2**. The metals that are marked with bold text are the ones that were selected.

Table 2 Measured environmental concentrations of seven metals in Göta älv and their respective toxicity data in rainbow trout (*Oncorhynchus mykiss*) for 28 days exposure

Substance	Environmental concentration[1] (µg/l)	LC_{50} (µg/l)	Toxic unit (Env conc/LC_{50})
Cd (µg/l)	0,02	130	0,00015
Cr (µg/l)	**1,2**	**170**	**0,007**
Cu (µg/l)	**2,3**	**90**	**0,025**
Hg (µg/l)	**<0,01**	**5**	**0,002[2]**
Ni (µg/l)	2,4	50	0,048
Pb (µg/l)	1,2	170	0,007
Zn (µg/l)	**9**	**170**	**0,053**

[1]Maximum concentration of seven metals measured at Alelyckan during the year of 2006.
[2]The toxic unit for mercury was calculated based on the worst case scenario that the concentration is 0,01 µg/l.
Source: Göta älvs vattenvårdsförbund, 2007; U.S. Environmental Protection Agency, 2000-2009.

Zinc was selected since it had the highest ratio and due to its known toxicity. Copper was selected since it can be very toxic to aquatic organisms and since a relatively high toxic unit was obtained. Chromium was selected based on the relatively high toxic unit, the fact that it can be very toxic and that there are a lot of chromium fillings in the Stallbacka area in Trollhättan. Mercury was chosen even though the toxic unit was low; the reason being its ability to biomagnify in the form of methylmercury. Nickel was not selected for further studies.

Pharmaceuticals constitute a diverse group of stressors that can give rise to many different effects. Pharmaceuticals can be divided into functional classes in terms of use. In **Table 3** below some of the major functional classes and the associated principal compounds of environmental concern are presented. The substances marked with bold text are the ones that were selected for further study. In total, four substances that are commonly used in Sweden[i] and that belong to different functional classes were selected. The selected substances were pharmaceuticals that to a low extent are "treated" in the WWTPs and those that have been proven to cause ecological effects at low concentrations (Bendz *et al*., 2005). The selected pharmaceuticals only include human pharmaceuticals. Veterinary pharmaceuticals are not included due to the current limited amount of data available regarding release of these substances to the environment.

[i] Apoteket Kundcentrum, e-mail conversation on the 19th of December 2008.

Table 3 Pharmaceutically active compounds and their uses

Compound group/class	Compound
Veterinary & human antibiotics	Trimethoprim, erythromycin, lincomycin, sulfamethoxazole, chloramphenicol, amoxicillin
Analgesics & non steroidal anti-inflammatory drugs (NSAIDs)	Ibuprofen, **diclofenac**, fenoprofen, acetaminophen, naproxen, acetylsalicylic acid, ketoprofen, indometacine, paracetamol
Neuroactive compounds (antiepileptics, antidepressants)	Diazepam, **carbamazepine**, fluoxetine, primidone, salbutamol
Blood lipid lowering agents	Clofibric acid, gemfibrozil, bezafibrate, fenofibric acid, etofibrate
β-blockers	Metoprolol, **propranolol**, timolol, sotalol, atenolol
X-ray contrasts	Iopromide, iopamidol, diatrizoate
Steroids & hormones	Estradiol, **ethinylestradiol**, estrone, estriol
Cytostatic compounds and cancer therapeutics	Methotrexate, tamoxifen, ifosfamide

Source: Ellis, 2006 and Fent *et al.*, 2006

Two substances were selected to represent the pesticide group, namely MCPA and diflufenican. Common for these two substances is that they emitted in large quantities, are the most frequently occurring pesticides in Swedish surface waters and that the measured concentrations often are close to or above the guideline values (Adielsson and Kreuger, 2008).

The organic pollutants were represented by the three groups of substances PAH (polycyclic aromatic hydrocarbons), HBCD (hexabromocyclododecane) and PFOS (perfluorooctyl sulfonate). These three substances have been identified as being of environmental concern. PAH can be highly toxic and carcinogenic (Hoffman *et al.*, 1995). HBCD and PFOS have been shown to be persistent and the knowledge about the ecological effects that they cause is largely unknown (Naturvårdsverket, 2008b). At the same time, measurements show that the emitted quantities of these two substances are increasing.

A summary of all of the selected stressors for each stressor group and a comment on why they were selected are shown in **Table 4** below.

Table 4 The selected stressors and the selection criteria

Stressors	Criteria
Metals:	
Copper (Cu)	Emitted in large quantities, toxic
Zinc (Zn)	Emitted in large quantities, toxic
Chromium (Cr)	Possibly emitted in large quantities, toxic
Mercury (Hg)	Toxic
Pharmaceuticals:	
Diclofenac	Emitted in large quantities, potentially toxic, persistent
Propranolol	Emitted in large quantities, potentially toxic, persistent
Carbamazepine	Emitted in large quantities, potentially toxic, persistent
Ethinylestradiol	Emitted in large quantities, toxic, persistent
Pesticides:	
MCPA	Emitted in large quantities
Diflufenican	Emitted in large quantities
Organic pollutants:	
Polycyclic aromatic hydrocarbon (PAH)	Emitted in large quantities, persistent, toxic
Hexabromocyclododecane (HBCD)	Emitted in large quantities
Perfluorooctyl sulfonate (PFOS)	Emitted in large quantities

The aquatic ecosystem was selected to be the object in need of protection and therefore the object of study. The reason for this choice was that as water is transported to Göta älv from the drainage area, either as surface water in tributary flows or via groundwater transport, it collects and transports stressors from the surrounding areas. As a result, the aquatic ecosystem gets exposed to the stressors and is therefore at risk of being subjected to adverse effects.

4 Preliminary assessment

4.1 Method

The main process of the preliminary assessment consisted of gathering available data about the selected stressors and stressor sources as well as about the ecosystems that potentially had been or are exposed to the stressors. This part of the assessment was qualitative with the aim of gathering available data that could be used to guide the next step of the procedure.

Different types of literature were used in this process. To describe the existing stressor sources, reports containing general information about the Göta älv river basin and the activities that are performed in the area were used. To describe the stressor characteristics, more specific information regarding chemical properties and toxicity data were used, with the aim of describing environmental fate, bioavailability, tendency for bioaccumulation etc. The expected ecological effects were also obtained by using toxicity data and information about the ecosystems. The ecosystems potentially at risk of experiencing adverse ecological effects were identified by looking at the stressor characteristics, which indicate where the stressors will end up in the river basin and which ecosystems that can therefore be exposed.

4.2 Results

The results of the preliminary assessment consist of four different parts; existing stressor sources, stressor characteristics, expected ecological effect and ecosystems potentially at risk, which is presented below.

4.2.1 Existing stressor sources

In this section, each of the selected stressor sources is described including which type of stressors they emit, the pattern of release and the location of the stressor source.

Industries, contaminated sites and landfills

Industries, contaminated sites and landfills can have a substantial effect on the environment due to the stressors that they emit. Examples of types of stressors are heavy metals, organic pollutants and nutrients. In order to get an overview of the number of industries, contaminated sites and landfills within the river basin of Göta älv, two directories from Länsstyrelsen were used in this study. These are the EMIR- and the MIFO-registers.

The EMIR directory registers businesses that perform activities that are harmful to the environment. That is activities that use of land, buildings and facilities that in some way result in emissions to air, groundwater, surface water or soil that can be harmful to human health or the environment (Naturvårdsverket, 2007). Each year, all businesses registered in the EMIR directory must hand in an environmental report in order to pursue their commercial operations. In the directory, information regarding location of the business, conditions of the permit, emissions etc are also kept.

The second directory, MIFO, is used to register contaminated sites. An inventory is currently being performed in order to determine how many contaminated sites there are in Sweden. This inventory is performed according to the MIFO procedure, a method for inventory of contaminated sites developed by the Swedish environmental protection agency. In the MIFO inventory, the different contaminated sites are ranked from 1-4 based on the risk that they

pose. The sites with highest risk are ranked 1 and the ones with the lowest risk 4. The inventory in Västra Götaland is expected to be finished by year 2013.

Landfills are included in both the EMIR directory and in the MIFO inventory. Here, landfills are treated as a separate source in this study since the stressors that they emit differ slightly from the industries and the contaminated sites.

Industries

Most industries in the Göta älv river basin are located in connection to the communities, with the exception of the municipality of Ale where there are industries situated along the entire river stretch. The most common industries located in the river basin today are chemical and engineering industries. Previously, also shipyards and pulp and paper industries were common along the river (Länsstyrelsen Västra Götaland, 2002). Studies on emissions to water have shown that the industries mainly emit/emitted metals and organic pollutants to the river water via waste water and cooling water (Johansson and Skrapste, 2003). The industries mainly emit stressors during operating hours since they are emitted via process wastewater and sewage, which are generated during operations.

According to Wolme[i], there are 79 EMIR-facilities registered within the study area. Among these facilities, there are chemical industries, metal and alloy industries, mechanical industries, pulp and paper industries, power generation facilities, landfills, wastewater treatments plants etc. A complete list of the EMIR-facilities can be found in Appendix A.

Contaminated sites

The contaminated sites within the study area have mainly been contaminated by past anthropogenic activities such as industrial activities, petrol stations and dumping of waste. Stressors are mainly emitted from contaminated sites via leaching through the soil profile followed by further leaching to the groundwater or leaching directly to surface waters. Studies have shown that the most common stressors that are emitted from contaminated sites in the Göta älv river basin are halogenated hydrocarbons (dioxins), oils, PAH and metals (Gohari, 2006). In contrast to the industries, the emissions from the contaminated sites are continuous. Another process that gives rise to emissions of stressors from contaminated sites is a landslide.

The current MIFO inventory results show that many of the industrial lots in the Göta älv river basin are possibly contaminated (Länsstyrelsen Västra Götaland, 2008). In the entire region of Västra Götaland, there are by estimate 10 000 potentially contaminated sites. According to Schultz[ii], there are 142 confirmed contaminated sites ranked as being of risk class 1 or 2 within the study area that are registered in the MIFO database. These MIFO objects include closed landfills, industrial lots, old gas stations, shipyards etc. For a complete list of all the MIFO objects, see Appendix D.

[i] Sten Wolme, responsible for EMIR facilities at Länsstyrelsen Västra Götaland. Excerpt from the EMIR directory obtained via e-mail on the 22nd of April 2009.

[ii] Uffe Schultz, responsible for inventory and supervision of contaminated sites at Länsstyrelsen Västra Götaland. Excerpt from the MIFO database obtained via e-mail on the 3rd of February 2009.

Landfills

Both active and closed landfills emit stressors to the environment. What is emitted from landfills is to a large extent affected by the type of waste that is contained within the landfill. Typical stressors are metals, organic pollutants and nitrogen. The stressors are mainly emitted via leachate. To reduce the environmental effects from landfills, the leachate is for some landfills collected and treated before being discharged into the environment. Today, there are a few landfills within the river basin of Göta älv that are still operating. Some of these landfills have their leachate water connected to WWTP, thereby reducing the environmental pressure from the landfill. In this study, the closed and operating landfills that do not have leachate treatment were considered as stressor sources.

Wastewater treatment plants

Wastewater treatment plants (WWTPs) receive wastewater from a range of sources such as households, industries and hospitals. The treatment plants clean the water by using different treatment steps. The most common treatment steps are mechanical, chemical and biological treatment. Mechanical treatment consists of grids that remove large debris and solid waste from the water. Chemical treatment uses chemicals to precipitate unwanted compounds and thereby removing them from the soluble fraction of the wastewater. Biological treatment uses bacteria to degrade chemical substances that are difficult to remove with the two previous treatment steps. After treatment, the effluent is discharged into the environment. The existing Swedish WWTPs are mainly focused on removing metals and nutrients from the wastewater since those were the main pollutants in the wastewater when the WWTPs where constructed. Today, the widespread chemical use within the society has led to an array of new stressors that are being emitted to the environment via wastewater, for example flame retardants and surface-active substances. These substances are not efficiently removed by the WWTPs and are therefore being emitted without treatment into the environment.

Some of the main stressors in wastewater today are metals, nutrients, bacteria, viruses and parasites, pharmaceutical residues and organic pollutants. In this study, the focus is on chemical stressors, i.e. pharmaceutical residues, nutrients, organic pollutants and metals. Since the treatment plants are operative during the entire day, the stressors from wastewater are constantly being released.

The majority of all the sewage in the study area (95 percent) is treated in the municipal wastewater treatment plants, while the remaining 5 percent is discharged through on site sewage systems (Åström and Pettersson, 2007). The on site sewage systems are not included in this study. There are seven municipal wastewater treatment plants located within the study area, which discharges their effluent into the Göta älv. In total, these treatment plants treat sewage from approximately 90,000 person equivalents (pe). In **Table 5** below, all the wastewater treatment plants within the study area are presented along with the number of connected people (personal equivalents) and the volume of treated and untreated wastewater. The untreated wastewater discharged from the wastewater treatment plants is often discharged during rainy days when there is not enough capacity to treat all the incoming wastewater or during operational disturbances in the treatment process.

Table 5 Wastewater treatment plants located in the Göta älv river basin between Vänersborg and Säveån and their respective number of connected people and volume of treated wastewater.

Municipality	Wastewater treatment plant	Connected people (pe)	Effluent flow per year (1,000 m^3)	Untreated effluent flow per year (1,000 m^3)
Vänersborg	Holmängen	27,000	4,900	1
Trollhättan	Arvidstorp	47,000	10,000	0
Lilla Edet	Nygård	500	33	-
	Lödöse	1,600	160	12
	Hjärtum	380	52	-
	Ellbo	6,200	1,400	97
Ale	Älvängen	6,400	790	43
Kungälv	Diseröd	1,100	130	0,8

Source: Åström and Pettersson, 2007

Agriculture and husbandry

Within the river basin of Göta älv, approximately 20 percent of the land (306 km^2), is used for agriculture (SMHI, 2008). A large part of this arable land is situated close to the shores of Göta älv, due to the fertile soil in the river valley. The agriculture affects the environment in many different ways, both positively and negatively. The agriculture helps preserving the diversity of small biotopes found in the agricultural landscape (Statistiska centralbyrån, 2005). At the same time, the use of fertilizers can cause eutrophication of nearby waters and the use of pesticides as well as pharmaceuticals can give rise to toxic effects in organisms. Since agriculture is a diffuse source, it is difficult to quantify the amounts of stressors that are emitted (Naturvårdsverket, 2008d).

Fertilizers are mainly used to keep the arable land from becoming nutrient poor as a result of natural leakage of nutrients from the soil. Nitrogen and phosphorous are the main nutrients used. When nutrients are leached from soil, they may cause eutrophication of nearby waters. Pesticides are used within the agriculture to reduce the amounts of weed, vermin and fungi that can negatively affect the yield of the harvest. The pesticides can be divided into three different subgroups, herbicides, insecticides and fungicides. When the pesticides are sprayed on the fields, there is a risk that they end up in the environment due to transportation by wind drift, surface run-off and leaching through the soil.

4.2.2 Stressor characteristics

The characteristics of the different stressors influence in which environmental matrix the stressor will end up, which type of toxicity that it can cause etc. When characterising each stressor, it can be good to base the characterisation on a set of common factors. The following factors were used in the characterisation:

- Type of substance
- Use
- Environmental fate
- Toxicity data

There are some things that are common for all stressors. The distribution of stressors in the environment is determined by factors such as the properties of the substance and the soil, the topography of the treated area, the climate, how it is used etc. (Törnqvist *et al*., 2002). The properties of the substance determine among others if the substance will dissolve in water and subsequently be transported by water or if the substance will bind to soil particles and sediment. The properties of the substance also determine the sorption properties, i.e. how hard the substance can bind to soil particles, which in turn affect the leaching of the substance. The

structure of the soil determines the rate of transportation through the soil as a result of the pore size, large pores cause rapid transportation. The composition of the soil determines the strength of the soil, if it easily comes apart or not. Calciferous soil easily crumbles and thereby releases aggregates of soil particles, which could be transported to other environmental compartments.

In the following sections, the different stressors are described in more detail with respect to the factors mentioned above.

Pharmaceuticals

Pharmaceutically active compounds constitute a diverse group of stressors that can give rise to many different effects. Current monitoring studies conducted worldwide show that it is common to find measurable concentrations of drug residues in treated sewage water as well as in surface water (Fent et al., 2006). Due to the pharmaceutical's intended application, these substances are often potent and persistent substances and they can give rise to many unintended effects when released into the environment. The main sources of pharmaceutically active compounds are sewage effluent and direct discharge from agricultural areas to receiving waters. These pathways are the principal ones, since the main use of pharmaceuticals is within human and veterinary applications (Bendz *et al.*, 2005). As mentioned previously, veterinary pharmaceuticals are not included in this study due to the limited amount of data regarding emissions of these substances to the environment.

When pharmaceuticals are used for human applications, the main route of entry to the environment is via ingestion, metabolism, excretion through faeces and urine, and finally disposal into the sewage. The excreted pharmaceuticals generally consist of a mixture of pharmaceuticals in native forms and as metabolites. In treated sewage water, it is possible to see a decrease in concentrations of pharmaceuticals when comparing influent and effluent sewage water, although, the removal efficiencies for pharmaceuticals vary a lot depending on substance and treatment plant (Bendz et al., 2005). Several compounds survive treatment at concentrations higher than half the influent concentration. Some of the compounds even showed an increased concentration after treatment as a result of cleavage of conjugates. Generally, pharmaceuticals that are not readily degraded get discharged. Due to the constant release of pharmaceuticals to the aquatic environment, the aquatic organisms become subjects to long-term exposure of these pharmaceuticals, which can cause both acute and chronic effects.

The behaviour and fate of pharmaceuticals in the environment are not well known today, although some facts are known. Generally, most pharmaceuticals have a low volatility, and therefore distribution is believed to occur primarily through aqueous transport, but also to some extent by food chain dispersal (Fent *et al.*, 2006). To determine the fate of different pharmaceuticals, the pharmaceuticals are often divided into subgroups depending on chemical properties, for example acidic and basic pharmaceuticals and zwitterions. Acidic pharmaceuticals are generally charged (polar) at neutral pH and tend to remain in the aqueous phase while basic pharmaceuticals and zwitterions often adsorb to particles and sludge to a significant extent in water, due to their neutral charge at pH around seven.

In natural waters, it is believed that the most important removal processes of pharmaceuticals are abiotic transformation reactions such as photolysis. The efficiency of the photolysis depend on factors such as substance properties, but also factors such as strength of solar irradiation which is affected by latitude and season, as well as constituents present in the

water that can act as photosensitizers by forming radicals. Besides abiotic processes, biotic processes also occur. Biotransformation through biodegradation is one, but the importance of this process is believed to be minor in comparison to the abiotic processes. Adsorption and desorption as well as cleavage of conjugates can also occur, which can affect the concentrations of pharmaceuticals in natural waters. Regarding bioaccumulation, there is almost no information available on pharmaceuticals in biota or food webs (Fent *et al*., 2006). The lifetime and persistency of the different pharmaceuticals is mainly determined by the loading rate relative to the collective rate of transformation. If the loading rate is higher than the transformation rate, the pharmaceutical will become persistent (Bendz *et al*., 2005).

Fent *et al.* (2006) have reviewed different studies to compile data about ecological toxic effects caused by pharmaceuticals and have shown that reproductive effects is a common effect in fish caused by many different classes of pharmaceuticals. Examples of pharmaceutical classes that can cause reproductive effects are steroids and hormones, NSAIDs, β-blockers as well as neuroactive compounds. Other chronic effects in fish are effects on the cardiovascular system (e.g. propranolol), oxidative stress and impairment of renal and gill function (e.g. diclofenac). However, chronic effects caused by pharmaceuticals are not fully understood, since traditional toxicity tests have been focused on acute toxicity performed in a few standard species (Fent *et al*., 2006).

In the sections below, there is some specific information about the use of the different substances as well as their environmental fate.

Diclofenac

Diclofenac is an anti-inflammatory drug that belongs to the group of non-steroidal anti-inflammatory drugs (NSAID). Diclofenac is an acidic pharmaceutical and it exists as negatively charged ions at neutral pH (Fent *et al*., 2006). Consequently it mainly occurs in the dissolved phase in wastewater. Though, if the pH is lowered, adsorption to solids increases. Generally, the removal of diclofenac in WWTPs is low. Two Swedish studies showed removal efficiencies of diclofenac to range between 22-25 percent (Bendz *et al.*, 2005; Paxéus, 2004). Other studies indicate removal efficiencies ranging from 17-69 percent (Bendz *et al*., 2005). A recent chronic ecotoxicological study of diclofenac indicate that diclofenac can cause histopathological changes in fish, especially to renal and gill tissue (Fent *et al*., 2006).

Propranolol

Propranolol is a β-blocker that is used in the treatment of high blood pressure as well as in the treatment of patients that have experienced a heart attack in order to prevent further attacks. Propranolol is a polar substance that is commonly found in the soluble phase of wastewater (Fent *et al*., 2006). A Swedish study indicates removal efficiencies of approximately 32 percent (Bendz *et al*., 2005), while a German study indicates a removal efficiency of 96 percent (Fent *et al*., 2006). Toxicity tests indicate that propranolol can cause chronic toxicity to both the cardiovascular system but it can also affect reproduction in fish (Fent *et al*., 2006).

Carbamazepine

Carbamazepine is a neuroactive compound that is mainly used as an antiepileptic since it reduces convulsions. This compound is a neutral compound with regard to pH that is polar and therefore soluble in water (Fent *et al*., 2006). It has been proven to be very persistent and that is poorly eliminated in WWTPs. A Swedish study show removal efficiencies around 30 percent (Bendz *et al*., 2005) while other studies show indicate removal efficiencies ranging

from 7-53 percent (Fent *et al*., 2006). Information regarding toxic effects in fish caused by carbamazepine is limited.

Ethinylestradiol

Ethinylestradiol is a steroidal hormone that is commonly used in contraceptive pills. This substance is hydrophobic with a log K_{ow} of 4.0 (Fent *et al*., 2006) and consequently it can adsorb to sludge and organic matter to a significant extent. Ethinylestradiol is therefore removed to a significant extent in WWTPs. Studies show removal efficiencies of approximately 85 percent (Fent *et al*., 2006). Besides the removal of ethinylestradiol in WWTPs, ethinylestradiol can also be generated in WWTPs as a result of cleavage of its conjugated metabolite. This process can therefore increase the concentrations of ethinylestradiol in wastewater. Even though ethinylestradiol is only emitted in low concentrations from WWTPs, the amounts that are emitted still have the potential to cause adverse effects to the environment. That is since ethinylestradiol can induce effects at very low concentrations. Endocrine disruptions in fish can be seen at concentrations in the range of ng/l. Toxic effects that are caused by ethinylestradiol mainly affect the reproductive system in fish due to endocrine disruption (Fent *et al*., 2006)

Nutrients

Nutrients are naturally occurring in the environment and they are essential for the growth of organisms. Nutrients are often divided into two different categories, macronutrients and micronutrients. The difference between the two groups is that macronutrients are substances that the organism requires in large quantities while micronutrients are only required in small or trace amounts. Here, the focus will be on the two macronutrients nitrogen and phosphorous. Since nutrients are essential, there is an optimal concentration of them in the environment. If the concentration is too low, growth will be limited and if the concentration is too high, there is a risk for eutrophication or toxic effects. Nutrients can also decrease toxicity of other substances. In agriculture, nitrogen and phosphorous are added to the land to avoid soils becoming nutrient-poor as a result of natural leakage of nutrients from the soil. The amount of nutrients that are leached from the agricultural lands is affected by many different factors. Among them is the amount of run-off, soil type, type of crop, type of fertilizer etc. As for contaminated soil, there are two main pathways for the leaching. The first one is leaching through the soil profile followed by further leaching to the groundwater. The second pathway is leaching directly to surface waters. The recipients for both these pathways are the natural waters. A study of the leakage of nutrients from arable land in Sweden (Naturvårdsverket, 2008d) showed that the leaching from the arable land in the Göta älv river valley is high compared to the rest of the country. This is in part correlated to the relatively large amount of precipitation in the area (approx. 730 mm/y)[i].

Nutrients such as nitrogen and phosphorus are generally the main limiting factors for primary production in aquatic ecosystems. Nitrogen is mainly limiting in marine ecosystems while phosphorous is limiting in freshwater ecosystems (Petersson, 2006). For Göta älv, phosphorous is therefore probably the main limiting nutrient. When nutrients are added to the aquatic ecosystems in high enough amounts, they can trigger an increased primary production, which in turn cause a chain-reaction of effects, collectively called eutrophication. For more detailed information about eutrophication, see Ærtbjerg *et al*. (2003).

[i] SMHI Kundtjänst, telephone conversation on the 14th of April 2009.

Since nutrients are released by a number of different sources, the pattern of release can vary to a large extent. Nutrient release from wastewater is generally only affected by external factors such as precipitation to a small extent. The release from agriculture and landfills on the other and is influenced by external factors to a larger extent (Ulén, 2008).

Nitrogen

Nitrogen can exist in different species. In water, the most common speciations are nitrate (NO_3^-), nitrite (NO_2^-), ammonium (NH_4) or in organic compounds. For ammonium to form anoxic conditions are required, otherwise nitrate or nitrite is formed instead. The toxicity of these different species varies; nitrite is known to inhibit oxygen transport while ammonia can work as a lung irritant (Timbrell, 2000). The transportation of nitrogen from soil to water is mainly dependent on the first pathway mentioned above, that is leaching through the soil profile followed by further leaching to the groundwater (Naturvårdsverket, 2008d). The average leakage of nitrogen from the arable land in the Göta älv river basin is 23 kg N/ha according to a study performed by the Swedish EPA (Naturvårdsverket, 2008d).

Phosphorous

Phosphorous can also exist in different species, but it mainly exists as phosphate (PO_4^{3-}) in the environment. In contrast to nitrogen, phosphorous can be transported from the soil via both of the two pathways mentioned above. It can both be leached through the soil profile and consequently get transported to the groundwater but it can also leach directly to surface waters (Naturvårdsverket, 2008d). The average leakage of phosphorous from the arable land in the Göta älv river basin is 0.94 kg P/ha according to a study performed by the Swedish EPA (Naturvårdsverket, 2008d).

Pesticides

As for the pharmaceuticals, pesticides are also a diverse group of stressors. Pesticides can be divided into three subgroups, herbicides, insecticides and fungicides. Even though the applications of these different types of pesticides are diverse, there are similarities between them. Pesticides are intended to target biologically active sites, thereby killing the pest. Pesticides are known many different modes of action. For example, herbicides can inhibit photosynthesis, prevent formation of essential amino acids and affect cell division and cell growth. Insecticides are known to work by affecting the nervous system, or by affecting protein synthesis and/or enzyme activity of the organisms that get in contact with the compound. A common mode of action for the third group of pesticides, the fungicides, is to inhibit the glycolysis and affect the protein synthesis. To conclude, pesticides can cause biochemical effects, pathological effects, cause behavioural changes and disturb growth, survival and fecundity impairment (Törnquist *et al*., 2002).

Many of these pesticides are designed in such a way that they are biologically degraded after treatment, thereby losing their biologically active function, but that is not always the case. For the substances that are not degraded, there is a risk that they could get dispersed into the environment and cause effects on other species than the intended ones. Since pesticides are targeted towards living organisms, they are potentially toxic to other living organisms as well. Studies made at different locations in Sweden show that detectable levels of pesticides are commonly found in surface water and ground water, sometimes above recommended guidelines (Ulén *et al*., 2002; Törnqvist *et al.*, 2002). These studies have also shown that it is most common to find pesticides in surface water during the summer months, which could be caused by a decreased dilution effect due to lower flows in the watercourses during this period.

Two herbicides have been selected to study here, MCPA (2-methyl-4-chlorophenoxyacetic acid) and diflufenican. These pesticides are the most commonly found pesticides in Swedish surface waters. Measurements show that the concentrations of these substances in surface water often are above the recommended guidelines and they can therefore pose a risk to the aquatic ecosystems (Adielsson and Kreuger, 2008).

MCPA

MCPA is an herbicide that inhibits growth in plant parts (Törnquist *et al*., 2002). After treatment with MCPA, the MCPA is degraded with a half-life of approximately seven days.

Diflufenican

Diflufenican is a selective herbicide that works when it comes in contact with monocotyledon and dicotyledonous plants (Swedish Chemicals Agency, 1997). It works by disturbing the plant production of beta-carotene, thereby inhibiting photosynthesis. The half-life of diflufenican in water is approximately 100 days. Diflufenican is lipophilic and is assessed to have a high potential to bioaccumulate.

Metals

Metals occur naturally in the environment, but they are also emitted by anthropogenic activities. Many metals are essential for the growth and survival of many organisms. If the concentration is too low, growth will be limited and if the concentration is too high, there is a risk for toxic effects. The background concentrations of for example chromium generally reflect the concentration in the underlying bedrock. In the society, metals are used widely for many different applications, especially in industrial applications. The emission of metals to the environment can occur either via emissions to the atmosphere as a result of combustion reactions followed by atmospheric deposition, or it can be released directly to the soil and waters via ware and tare of materials containing metals and via wastewater. During the last three decades, the anthropogenic emissions of metals have decreased. Today, the concentrations in running waters are generally much lower than the concentrations that were measured during the 1970s, when the concentrations generally were at its highest. In many running waters, the concentration of metals is now close to background concentrations.

Metals can occur in different species depending on abiotic conditions. In these different forms, the metals can give rise to different toxic effects. In the sections below, the speciation of the different metals considered in the study will be described in more detail together with some additional information.

Copper

Copper is a naturally occurring component in many minerals, and there is therefore a background concentration of copper in the environment. Copper is also an essential nutrient for many organisms. Within the anthroposphere, copper is used widely because of its high electrical and thermal conductivity. One of the major applications is within electrical equipment (Swedish Chemicals Agency, 2009). In Sweden, copper is also widely used as piping material. Relatively large amounts of copper are therefore emitted to the environment via WWTP. Copper can exist in different forms; in minerals copper usually exist in elemental form while free copper often exist as oxides or sulphides (CHEMnetBASE, 2009). In freshwater, the dominating oxidation state of copper is Cu(II) (Di Giulio and Hinton, 2008). A part of these ions will be present as a hydrated cation, which is often referred to as free copper. The free copper has a high affinity to ligands that occur in natural waters such as

carbonates, hydroxides, dissolved organic particles etc. As a consequence, dissolved copper in freshwater mainly exist as complexes with such ligands.

Studies have shown that physiochemical properties of the exposure water have a large effect on copper bioavailability and therefore also on the copper toxicity (Di Giulio and Hinton, 2008). The main toxic effect of copper on fish is disruption of the osmoregulation at the fish gills caused by decreased uptake of sodium in combination with increased sodium loss at the gills. Increased complexation of copper generally decreases the toxicity due to decreased uptake. The complexation is dependent on factors such as the ion content, pH and alkalinity of the exposure water. Besides these factors, calcium also affects copper toxicity by regulating permeability of the paracellular junctions where the sodium is lost (Di Giulio and Hinton, 2008). The disrupted osmoregulation can in turn cause reduced growth and pathological changes. Toxicity test have shown that the early life stages of fish are most sensitive to copper toxicity (CHEMnetBASE, 2009).

Zinc

As for copper, zinc is a naturally occurring metal and an essential nutrient. Zinc is very commonly used in alloys to enhance the properties of steel. Zinc is also used in batteries (CHEMnetBASE, 2009). One of the most important release mechanisms of zinc to the environment is through corrosion. The corroded zinc can then enter the environment via storm water flows and sewage. Close to smelters and galvanisation facilities, there can be very high contamination of zinc in both soil and water (Swedish Chemicals Agency, 2009).
Zinc toxicity mainly cause injuries to the gill tissue in fish. This toxicity has been shown to be dependent on temperature and hardness of the exposure water. High temperatures seem to increase elimination rates, at the same time increased hardness appears to reduce uptake of zinc. The effects of hardness seem to be more related to changes is the gills rather than metal speciation of zinc (Hoffman *et al.*, 1995).

Chromium

Chromium exists in many different oxidative states; for example as elemental form, as chromium(III) and as chromium (VI). The elemental and trivalent forms are the most commonly naturally occurring states, while the majority of hexavalent chromium is anthropogenic. The toxicity of the different oxidative states is different. Chromium(VI) is classified as more toxic that the other oxidative states. Chromium compounds are generally used in ferrochrome production, electroplating, pigment production and tanning. These industries, as well as the burning of fossil fuels and waste incineration are common sources of chromium in the environment. There is an environmental cycle for chromium, where the chromium cycles from rocks and soil, to water biota and air and after that back to the soil.

Toxicity tests exposing fish to chromium have shown that chromium can be both genotoxic and cytotoxic to fish cells, resulting in pathological changes and reduced growth (Goodale *et al*., 2008). However, the mechanism of chromium toxicity is still unknown. Fish exposed to chromium can also experience a condition decrease caused by reduced carbohydrate, protein and lipid content (Gangolli, 2005).

Mercury

Mercury is a metal that is virtually ubiquitous in the environment. Most elevated levels are though found close to anthropogenic sources. The most important sources of mercury in the environment are coal piles, solid waste and mine tailings. Leachate from tailings and landfills has also been proven to contain mercury. In the environment, mercury has a strong affinity to

organic matter, clays and hydrous metal oxides. In the water column, mercury is most commonly found bound to suspended solids or in the sediments. Mercury can occur in many different speciations in the environment. Besides occurring as ions with different valence, mercury can also exist as organometallic compounds, such as methylmercury. Methylmercury is formed by methylation, which can occur both biologically and chemically. Biological methylation is though the dominating process out of the two. Even though the concentration of methylmercury in the environment is low, it is still the most common form found in aquatic organisms. That is because methylmercury is rapidly accumulated in tissue and retained for long periods. Methylmercury have been proven to both bioaccumulate and to biomagnify (Swedish Chemicals Agency, 2009).

The behaviour of mercury changes drastically when transformed into its organometallic form, e.g. methylmercury (Di Giulio and Hinton, 2008). The toxic effects of methylmercury that have been noted are effects on the nervous system, liver toxicity as a result of lipid peroxidation, but also reduced lipid reserves in the liver (Drevnick *et al.*, 2008). Toxicity tests in juvenile fish indicate reduced juvenile survival due to impaired growth and immune function. Reproductive disturbances can also be linked to methylmercury exposure as a result of impaired testicular development in juvenile fish (Friedmann *et al.*, 1996).

Organic pollutants

Organic pollutants have been monitored in the Swedish environment for a long time, ever since the adverse effects caused by PCB (polychlorinated biphenyl) and DDT (dichlorodiphenyltrichloroethane) were first noticed. Today, the concentrations of these classical organic environmental pollutants are low and continue to decrease as a result of bans and regulations, and the risk that they pose to the environment has been significantly reduced (Naturvårdsverket, 2008b). Today, the current use of other organic compounds in the society causes new environmental concerns. Here, three substances have been chosen for further study, PAH, hexabromocyclododecane and perfluorooctyl sulfonate. These compounds are all emitted in large amounts into the environment, the concentrations are increasing and they appear to cause adverse effects on ecosystems (ref).

Polycyclic aromatic hydrocarbons (PAH)

Polycyclic aromatic hydrocarbons (PAH) belong to the group of aromatic hydrocarbons that contain two or more fused benzene rings. Attached to the rings, there can be different substituted groups, giving the PAH different properties. PAH can enter the environment from both natural and anthropogenic sources. Examples of natural sources are forests, grass fires and volcanoes. The predominant anthropogenic source of PAH is incomplete combustion of organic material. Examples of such sources are vehicles, residential heating, aluminium production, incineration and power generation. Another source is ware and tare of car tires since car tires contain highly aromatic oils that are rich in PAH. PAH have been found in all environmental matrices, i.e. the atmosphere, soil, water and biota (Yu, 2005). For surface waters, the most important sources are atmospheric deposition, municipal wastewater, urban storm water and industrial discharges.

The environmental fate of the PAH is largely affected by their properties. In general, PAH have low solubility in water, high melting and boiling points and low vapour pressure. The physical and chemical properties of PAH vary according to their molecular weight. As the molecular weight increase, the resistance to oxidation and reduction increases while vapour pressure and solubility decrease (Yu, 2005). PAH are also persistent, and the persistency increase with increasing molecular weight. When released into the atmosphere, the PAH have

a strong affinity to airborne organic particles. The particles can then be transported for long distances before they get deposited. The deposition occurs by both wet and dry deposition. After deposition, a large fraction of the particles containing PAH are transported to surrounding natural waters via storm water flows. In water, the PAH are known to bioaccumulate in the tissues of organisms or accumulate in the sediments due to its high lipophilicity. In the organisms, the PAH can give rise to many different toxic effects. Some PAH are e.g. carcinogenic (Yu, 2005).

Other effects that can be caused by PAHs are endocrine disruption, reduced growth and biochemical changes. The carcinogenetic effects arise from reactive metabolites that form during the metabolism of PAH. The larval and juvenile stages of fish are extra sensitive to PAH. Reduced growth and cellular abnormalities have been shown be effects of PAH exposure during these early life stages. The toxicity of PAH seems to increase with increasing molecular weight and it is also affected by the concentration of dissolved oxygen (Hoffman *et al.*, 1995).

Hexabromocyclododecane (HBCD)

Hexabromocyclododecane (HBCD) belongs to the group brominated flame-retardants. The main application of HBCD is to reduce the flammability of polystyrene that is used in buildings for thermal insulation (Cheaib *et al.*, 2009). Some HBCD is also used in furniture upholstery. HBCD is one of the most commonly used brominated flame-retardants and is produced in large volumes each year. The use of HBCD has increased during recent years as a result of the ban of penta- and octa-BDE (bromodiphenylether). During application of the HBCD to the product, there is a risk of volatilization into the atmosphere. Other sources of HBCD are WWTPs and landfills. Today, HBCD is ubiquitous in the nature, and the environmental concentrations are increasing. HBCD is persistent and lipid soluble. Studies have shown that HBCD can bioaccumulate and biomagnify (Morris *et al.*, 2004).

For HBCD, the toxic effects are largely unknown. There are some studies that indicate that HBCD may induce cancer via non-mutagenic mechanisms (de Wit, 2002).

Perfluorooctyl sulfonate (PFOS)

Perfluorooctyl sulfonate (PFOS) is formed by metabolic breakdown and environmental degradation of perfluoroalkylated substances (PFAS). The PFAS are used within a wide range of applications due to their special chemical properties; they are both lipophobic and hydrophobic (Bossi *et al.*, 2008). The main applications for PFAS are surface treatment and fire-fighting foams. A large fraction of the PFAS that are used is emitted to the environment via wastewater effluent as a result of ware of products (textiles, carpets, leather, paper etc.), but they are also emitted from landfills (Hekster *et al.*, 2003). During wastewater treatment, the concentrations of PFOS have been shown to increase. This effect is probably caused by degradation of the precursor PFAS during the treatment (Bossi *et al.*, 2008). Another potential source of PFOS in the Göta älv river basin is paper and pulp industries, due to the use of PFOS for paper and board protection. Today, PFOS is detected in water, sediments and biota worldwide. During the past decades, the concentrations of PFOS in the environment has increased and is today of environmental concern (Naturvårdsverket, 2008b).

PFOS is a highly soluble and persistent compound that is not metabolized in biota. Studies have shown that PFOS bioaccumulates in fish, with a bioaccumulation factor ranging between 2,796-125,000 (Hekster *et al.*, 2003; Bossi *et al.*, 2008). PFOS has also been shown to be subject to biomagnification, with a biomagnification factor of 5.88 (Martin *et al.*, 2004).

Measurements of environmental concentrations of PFOS have shown that sediments appear to contain the highest concentrations of PFOS, but water also contains measurable concentrations (Martin *et al.*, 2004).

The knowledge about the toxicity to aquatic organisms caused by PFOS is today not fully known, especially not chronic toxicity. Studies have shown that PFOS is moderately acutely toxic and slightly chronically toxic to aquatic organisms. Toxic effects that are known are peroxisome proliferation in liver cells and enzyme induction. Other potential effects are carcinogenicity, immunotoxicity and reproductive and developmental effects (Bossi *et al*., 2008).

4.2.3 Expected ecological effects

Ecological effects can be caused separately by the different stressors, but also as a result of a mixture of the stressors. For stressors that target the same pathways, a mixture can contribute to decreasing the concentration at which negative effects by the individual stressors are expected. Exposure to a mixture of stressors can also decrease the effects of one another as a result of opposite effects. In the sections below, the individual toxic responses from the stressors will be summarised, but there will also be some information about toxic effects of mixtures.

Effects caused by toxic response

As described in the section about stressor characteristics, the selected stressors are able to induce toxic responses in organisms, which in turn can result in adverse ecological effects. As is also described, the knowledge about ecological effects caused by some of the stressors is deficient; for example effects caused by HBCD and some pharmaceuticals. In **Table 6** below, the toxic effects caused by the different stressors are summarised. All of these effects can be classified as being on an organism level. In the table, it can be seen that pathological effects is the most common response. Pathological changes can though occur without causing any negative effects to the health of the organism. Thus pathological changes do not necessary cause adverse ecological effects.

Table 6 The selected stressors and their potential toxic effects on fish

Stressors	Biochemical effects	Pathological effects	Condition decrease	Growth, survival and fecundity impairment
Pharmaceuticals	•	•	•	•
Pesticides and biocides	•	•		•
Metals:				
Cu	•	•		•
Zn	•	•		
Cr		•	•	•
Hg		•	•	•
Organic pollutants:				
PAH	•	•	•	•
HBCD[1]		•		
PFOS	•	•	•	•

[1] Toxic effects caused by HBCD are largely unknown and the quality of the performed studies is under question.

The adverse effects at the organism level described above can be reflected at the population level as reduced reproduction and an abundance decrease caused by a reduced general condition of the fish population. Effects on the population level can then progress up to the

community level, resulting in reduced density and diversity of species, as well as trophic structure changes.

Effects caused by eutrophication

As mentioned above, the cause of eutrophication is increased loadings of nutrients, e.g. nitrogen and phosphorous, to the aquatic ecosystems originating from agriculture, WWTPs, transportation, landfill leachate etc. Increased concentrations of nutrients in the aquatic ecosystems can trigger an increased primary production. An increased abundance of phytoplankton increases the abundance of primary consumers such as zooplankton, which in turn increases the available food for secondary consumers such as fish, causing an increase in the fish population. Besides triggering primary production, increased nutrient concentrations can also lead to decreased concentrations of silica in the water, favouring plankton that does not need silica to grow (Lundberg, 2005), i.e. e.g. green algae are favoured versus diatoms. As a result, the composition of plankton species can change. An increased production of phytoplankton can cause increased sedimentation of organic matter, which in turn affects both the transparency of the water and the oxygen consumption. A decreased transparency can cause changes in the benthic flora, which negatively affects the living conditions for zoobenthos and as a consequence, fish habitats are reduced (Lundberg, 2005).

In summary, eutrophication can cause an increased production of both flora and fauna, but the species diversity is often decreased due to changed living conditions. The main effects of eutrophication will not be seen in running waters, but in the oceans since running waters are less sensitive to eutrophication due to the constant flow washing the nutrients away.

Effects caused by mixtures of stressors

Besides the individual effects that a stressor can cause, a mixture of stressors can cause additional effects. However, the toxicity of mixtures of stressors is still to a great extent unknown, since standardised toxicity tests do not acknowledge these interactions.

The effects caused by the toxic effects of pharmaceuticals, pesticides and other compounds, can also be masked by the effects of eutrophication. Population decreases as a response to the toxic effects can be evened out due to population increases as a result of increased primary production caused by eutrophication. However, the main effect of the eutrophication will not occur in the river but in the estuaries. However, the extent of this effect is unknown.

4.2.4 Ecosystems potentially at risk

In the planning phase, the aquatic ecosystem of Göta Älv was identified as being particularly susceptible to adverse ecological effects as a result of the stressors being transported to the aquatic matrix via different pathways.

Within the aquatic ecosystem in the river basin of Göta Älv, the risk to the different habitats varies. Habitats in close connection to the discharges of the stressor sources are more likely to experience adverse effects than other habitats, since the stressors have not yet been diluted close to the source. Studies have shown that complete mixing of an emission to Göta älv occurs first after the water has moved 10 km downstream of the site of emission (Åström and Pettersson, 2007). If there are sites where there are many sources located close together, there is a risk that the ecosystem in this area is exposed to stressor concentrations that are much higher than the general concentrations after dilution have taken place in the river.

4.2.5 Summary of preliminary assessment

The preliminary assessment showed that there are many stressor sources within the study area that can potentially cause adverse effects to the environment. Of these stressor sources, all emit more than one stressor, which makes the spatial and temporal distribution of emissions complex. As water is transported within the drainage area, it collects and transports stressors from the surrounding areas to Göta älv, thereby making the aquatic ecosystem in the river one of the endpoint matrices of these stressors in the environment. Looking at the total emissions from the different sources, Göta älv receives a constant input of stressors that all individually have the potential of causing adverse effects. The characterization of the stressors yielded that many of the stressors in this study are persistent, potent and also have the ability to bioaccumulate and biomagnify in the higher trophic levels. The stressors can therefore pose a risk to the ecosystem. In the regional risk assessment, more detailed investigations have been made in order to assess the environmental risk in the study area. The results from this study will be presented in the following sections.

5 Regional risk assessment

5.1 Method

The regional risk assessment was divided into three different phases; problem formulation, the analysis phase and risk characterisation. In this study, the risk was characterised with regard to subareas within the studied region, but also with regard to the different stressors.

5.1.1 Problem formulation

The problem formulation consisted of three different parts, selection of the assessment endpoints, summarising the problem in a conceptual model and development of an analysis plan for the assessment of exposure and effects at a regional scale. How the different parts were performed is described below.

Assessment endpoints

The assessment endpoint in an ERA is the environmental value that is to be protected (Moraes, 2002; US EPA, 1998). This endpoint can be defined at any level of biological organization. There are certain aspects to consider when selecting an assessment endpoint for a study. The assessment endpoint should be ecologically relevant, be susceptible to the stressors considered in the study and it should be included in management goals and have a high social value (Moraes, 2002). All of these aspects were considered when choosing an appropriate assessment endpoint for this study.

Conceptual models

To visualize how the emission of stressors is related to environmental effects at different levels, conceptual models can be used. These conceptual models represent simplifications of the real world with the aim of pointing out the most central aspects. A conceptual model showing sources of emissions and environmental fate of the stressors were developed.

Analysis plan

The analysis plan describes how the different assessments in the analysis phase should be conducted. The assessments in the regional risk assessment were based on information available in literature, as for the preliminary assessment. More specifically, the analysis plan should describe how to perform the exposure effect assessments.

Exposure characterisation

The exposure characterisation consisted of two parts. The first part aimed at assessing where the stressor sources and the habitats of the assessment endpoints are located. Hydrological data from SMHI was used to divide the study area into smaller subareas. For each subarea, the density of stressor sources and habitats was calculated. The information regarding the stressor sources and their geographical distribution was mainly obtained from Schultz and Wolme at Länsstyrelsen Västra Götaland, see section 4.2.1 above. Additional data regarding information about small WWTPs and landfills was obtained from the municipalities in the river valley (Ale kommun, 2008; Kungälvs kommun, 2008; Vänersborgs kommun, 2008; Trollhättan Energi AB, 2009) and from Göta älvs vattenvårdsförbund (2006) respectively. Data about the distribution of agricultural land in the river valley was obtained from SMHI. Information

regarding salmon and trout habitats within the study area was obtained from Molander[i] at Länsstyrelsen Västra Götaland.

In order to evaluate the distribution and density of the sources, the gathered data was compiled using the GIS software ArcGIS®, which then allowed for calculation of the densities of stressors in each subarea. By placing the different layers on top of each other, high-risk areas, that is areas where both stressors are emitted and habitats exist, could be identified.

The second part of the exposure characterisation consisted of determining what the predicted environmental concentrations (PEC) of the stressors might be within the study area. According to TGD, both measured data and model calculations should preferably be used when deriving exposure concentrations due to the uncertainty in the assessment of exposure. When analysing the data, preference should be given to adequate and representative measured data when available. In this thesis, both measured data and model calculations were used. Measured data were obtained from scientific literature and published results from local measurement campaigns. However, site-specific data could not be obtained for all stressors. When site-specific data were missing, data measured in either Sweden or Denmark, were used as a substitute. To better account for the conditions in the Göta älv river valley, some data was re-calculated with respect to use and dilution of the stressors. For those stressors for which adequate and representative data are lacking, model calculations were also performed in order to compare with calculated PECs based on measured data from other locations. The substances for which the environmental concentration were modelled are all of the pharmaceutical compounds, the two pesticides and the organic pollutant PFOS. Model calculations would also have been preferred for PAH and HBCD, but due to lack of data these calculations could not be performed.

In order to perform the model calculations, a simple dilution model for the river basin of Göta älv was created. The model calculates how the concentrations of stressors change in the river, from Vänersborg to Göteborg, which is visualised in **Figure 5** below.

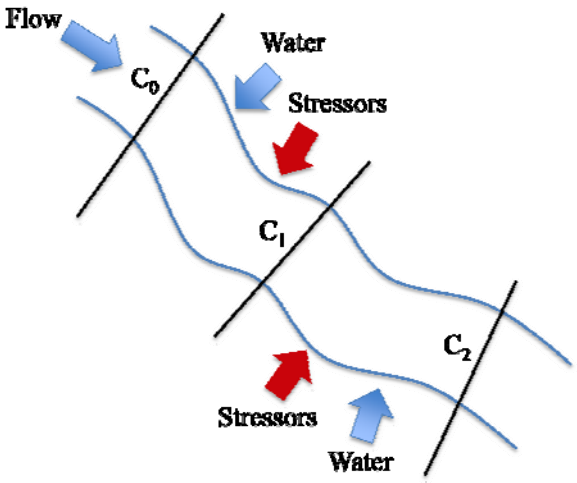


Figure 5 The PEC model

The study area was divided into subareas, based on the hydrological data from SMHI, so that the concentrations of stressors could be obtained at different locations along Göta älv. The

[i] Lars Molander, fishing adviser at Länsstyrelsen Västra Götaland, personal meeting on the 8th of April 2009.

water flow at Vänersborg, where Göta älv drains from Lake Vänern, was used as base flow, Q_0. As the river flows, water is added to the river from the different catchments due to precipitation and from wastewater treatment plants. For each subarea, the amount of added water was calculated based on the area of the subarea, average precipitation for the entire study area and how much of that precipitation that is expected to reach the river via surface water or ground water. For the catchment areas where there is a wastewater treatment plant located, the treated volume is added as well. Based on the original flow of Göta älv at Vänersborg and the amount of added water from each subarea, the cumulative of the river could be calculated. To account for seasonal variations in water flow and precipitation, three different scenarios were developed based on the water flow in the river:

1. High flow
2. Medium flow
3. Low flow

As seen in **Figure 5** above, stressors are also added to the river from the different subareas as the river flows from Vänersborg to Göteborg, as a result of transportation processes. For each subarea, the amount of added stressors was calculated based on the geographical distribution of stressors within the different subareas. Due to limitations in data and the complexity of the emission patterns, it was not possible to consider all relevant sources of emissions when calculating the loading of stressors from each subarea. In the sections below follows descriptions of which sources that are considered in the model.

The only source of pharmaceuticals considered in this model is WWTPs. Landfills can also emit pharmaceutical residues, but the emissions from landfills are expected to be much lower than the emissions from WWTPs. Emissions from landfills were therefore not considered. The predicted environmental concentrations were based on measured concentrations of pharmaceuticals in sewage effluent. The measured data was then recalculated to mass of emitted pharmaceuticals per person equivalent by multiplying the measured concentration with the annual flow of the WWTP and then dividing it with the number of connected pe. To get representative concentrations of pharmaceuticals for the WWTPs in the Göta älv river valley, the obtained mass per pe was then multiplied with the number of connected pe and divided with the annual flow of the same WWTP.

The pesticides are considered to be emitted from one source, that is agricultural land. The amounts of emitted pesticides were calculated based on the use of pesticides per agricultural area, the average leakage of pesticides from agricultural land and the area of agricultural land within each subarea.

The concentrations of the organic pollutant PFOS were calculated based on WWTPs being the only source. The emissions of PFOS from WWTPs were calculated in the same way as the emissions of pharmaceuticals. Landfills are also an important source of PFOS, but since there were no data regarding average leakage from the landfills within the study area, emissions from landfills could not be included in this study. Concentrations of HBCD could not be modelled due to lack of data. Concentrations in landfill leachate were found, but since no information on the average leakage was available no calculations could be performed. For HBCD in sewage effluent, there are no adequate data published today. The Swedish Environmental Research Institute, IVL, is currently performing measurements of HBCD in

sewage effluent. The data from IVL's new HBCD study is expected to be published during the year of 2009 according to Brorström-Lundén[i].

Based on the cumulative flow in the river and the loading of stressors from the different subareas, the cumulative concentration of stressors in Göta älv could be calculated according to the following equation,

$$c_i = \frac{cumulative\ mass\ (stressor)}{cumulative\ flow} = \frac{\sum_i m\ (stressor)_i}{\sum_i (Q_{river_i} + Q_{precipitation_i})} \quad \text{(Eq. 1)}$$

Where c_i is the concentration of a stressor at a specific subarea i, m_i is the mass loading of a stressor from a subarea (kg/month) and Q_i is the flow of the river (m^3/month) and the added water from precipitation. In total, the model generates concentrations calculated at 25 different locations along the river.

Effects characterisation

To be able to characterise the risk connected to each stressor, the potential effects need to be identified and at which concentrations they occur. For the effect characterisation, it is of interest to estimate dose/concentration-response relationships in order to assess the risk of adverse effect on the ecosystems in the river basin. Predicted no effect concentrations (PNEC) were therefore used. PNECs were obtained from scientific literature, such as reports from the Swedish EPA (Naturvårdsverket, 1999; Naturvårdsverket, 2008a; Naturvårdsverket, 2008c) as well as in a report by Ferrari *et al.* (2003) and the ECOTOX database provided by the US EPA (2000-2009).

Most of the PNEC values used in this study were derived from a report published by the Swedish Environmental Protections Agency (Naturvårdsverket, 2008c). In this report limit values for a set of environmental pollutants are proposed. According to the water directive, each country that is included by the directive must derive limit values for ecological effects in order to know at which concentrations adverse effects can be expected. To derive these limit values, PNECs have been estimated and these PNECs are the ones used in this study. From the report by Naturvårdsverket (2008c), PNEC values for MCPA, diflufenican, Cu, Zn, Cr, HBCD and PFOS were obtained. PNEC values for three of the pharmaceuticals; diclofenac, propranolol and ethinylestradiol were obtained from another report published by the Swedish EPA (Naturvårdsverket, 2008a). That study aimed at evaluating the wastewater treatment plants ability to treat pharmaceutical residues. Here, the PNEC value had been derived by FASS, Pharmaceutical Specialties in Sweden. FASS constitutes a compilation of information about pharmaceuticals obtained from the pharmaceutical companies. For the fourth pharmaceutical, carbamazepine, no PNEC was given in the report. Instead, data from a European study performed by Ferrari *et al*. (2003) was used. The PNEC values for Hg and PAHs were obtained from grounds for judgement on contaminated sites published by the Swedish EPA (Naturvårdsverket, 1999). Here, the grounds for judgement correspond to PNECs, which were derived from LC_{50}. Although, the assessment factors for these PNECs are high, indicating large uncertainties.

[i] Eva Brorström-Lundén, the Swedish Environmental Research Institute (IVL), phone conversation on the 19th of April 2009.

For some of the stressors, there is more than one PNEC value available reflecting different abiotic conditions. For Cu, a worst-case PNEC was derived based on the highest probable bioavailability in freshwater. For this PNEC to be applicable, the pH, alkalinity, DOC, iron and aluminium concentrations must be within a particular range. Comparisons to the conditions in Göta älv (Göta älvs vattenvårdsförbund, 2007) show that these guidelines are applicable to the study area. For zinc, two different PNECs were developed to account for different toxic effects as a result of the hardness of the water. In Göta älv, the higher value of the two PNECs was applicable since the hardness of the water is above 24 mg $CaCO_3$/l (Göta älvs vattenvårdsförbund, 2007).

5.1.2 Analysis phase

The results from the analysis phase present data on geographical distribution of stressor sources and habitats for the assessment endpoint, to show potential overlaps. Data on PEC and PNECs at these concentrations are also presented. These data are then used as a decision basis during the risk characterisation. The analysis phase was performed according to the analysis plan.

5.1.3 Risk characterisation

When the exposure characterisation and the effects characterisation have been conducted for all relevant environmental matrices, a risk characterisation is carried out. The risk characterisation can be either quantitative or qualitative dependent on the quality of the existing data sets. In this study, the risk characterisation was semi-quantitative since the quantitative data contained a lot of uncertainties. As mentioned, the risk will be characterised with respect to subareas within the studied region in order to identify high risk areas, but also high risk stressors.

Risk characterisation using the relative risk model (RRM)

The relative risk model (RRM) is a model that was developed by Wiegers and Landis in 1997 in order to generate regional risk hypotheses that could support the risk management process (Landis, 2005). In the model, the study area is divided into subareas, which are then given relative risk scores based on ranks of density of sources and density of habitats within the subarea and by applying weighting factors to these ranks. In this study, the relative risk model was used to identify high risk areas within the study area. Below, a more detailed description of the model, the ranks and the weighting factors can be found.

Risk can be obtained by integrating two factors, the likelihood of exposure and the likelihood of adverse effects to the endpoint. This is the foundation for the calculations used in the RRM (Moraes, 2002). In the RRM, three major factors are considered, density of stressors, density of habitats and effects parameter. These parameters are then converted into dimensionless numbers by applying ranking and weighting factors to the data. The model used in this study was the original RRM modified by Moraes (2002). The modified model also takes into consideration the relative amount the different sources emit the stressors, by using an extra weighting factor. Below, the different ranking and weighting factors used in this study are presented.

The ranking system for the density of stressors was based on the number of sources per area. For WWTPs and agriculture, a slightly different unit was used for the ranking. Since there are large variations in size between the different WWTPs in the studied area, they were instead ranked based on the number of person equivalents. For the agriculture, it is not possible to assign a discrete ranking factor such as number per area; instead the percentage of agricultural

land is used as the unit of the ranking factor. Ranks were given according to a two-point scale from 0 to 6. The different rank levels stand for:

- 0: no stressor sources
- 2: low density of stressor sources
- 4: medium density of stressor sources
- 6: high density of stressor sources

The intervals for the different densities of stressor sources were developed based on the subarea that had the highest number of stressor sources per area. The highest density of stressors in a subarea was used as the upper limit for the ranking intervals after having been rounded off to the nearest hundred. After that, the interval was divided into equally sized classes, representing the different densities. To get numbers that were easy to interpret, the densities were calculated per 1000 square kilometres. This method of ranking allows for relative comparison of the different subareas toward each other. The criteria for the ranking of stressors are shown in **Table 7** below.

Table 7 Criteria for scoring density of stressor sources, RS

Sources	No: rank 0	Low: rank 2	Medium: rank 4	High: rank 6
Wastewater treatment plants (pe per 10^3 km^2)	0	1-500	501-1000	> 1000
Industries (number per 10^3 km^2)	0	1-200	201-400	> 400
Contaminated sites (number per 10^3 km^2)	0	1- 400	401-800	> 800
Landfills (number per 10^3 km^2)	0	1-30	31-60	> 60
Agriculture (percentage of agriculture per area)	0	1-20	21-40	> 40

The weighting factor that accounts for differences in how much of the stressors the different sources emit was based on three different levels:

- 0: no/low emissions
- 1: medium emissions
- 2: high emissions

The weighting factors were assigned based on estimations of the relative amounts of emissions from the different sources for each group of stressors. Unlike the ranking of the stressors, there are no absolute criteria for how to assign the weighting factors for this parameter. The weighting factors were distributed in a way where the dominating source was given weighting factor 2, while smaller sources were assigned a factor 1. Sources from which there were no or insignificant emissions, weighting factor 0 was assigned. In **Table 8** below, all of the sources considered in study were assigned weighting factors for all of the different stressors.

Table 8 Weighting factors for stressors reflecting the relative amount of stressors emitted by each source, WS. Weights represent no/low (0), medium (1) and high (2) release of stressors from each source.

Sources	Pharmaceuticals	Pesticides	Nutrients	Metals	Organic pollutants
Wastewater treatment plants (pe per 10^3 km^2)	2	0	2	2	2
Industries (number per 10^3 km^2)	0	0	0	2	1
Contaminated sites (number per 10^3 km^2)	0	0	0	1	1
Landfills (number per 10^3 km^2)	0	0	1	1	1
Agriculture (percentage of agriculture per area)	0	2	2	0	0

The criteria for ranking density of habitats are based on expert judgement by Lars Molander[i] of where the habitats of salmon and trout are located within the study area. Subareas where there are no known habitats are given rank 0. Subareas where the endpoint resides are given rank 1 and subareas where there are important playgrounds are given rank 2, see **Table 9** below.

Table 9 Criteria for scoring density of habitats, RH

Habitat	No: rank 0	Low: rank 1	High: rank 2
Salmon and trout habitats	No habitats	Areas where the fishes reside	Important playgrounds

The final factor is the weighting factor for effects based on vulnerability of the endpoint to the different stressors. Here, factors such as toxicity, persistence and background concentrations have guided the selection of the ranks. The ranking scale goes from 0-6, with seven different possible levels, zero for not vulnerable and 6 for highly vulnerable. Pharmaceuticals and organic pollutants were given high ranks due to their general high toxicity. Pesticides have been ranked lower since the pesticides studied here have a short half-life. Also, they are not emitted during the entire year, which reduces the exposure. Nutrient and metals are both naturally occurring and have therefore been given relatively low ranks. Metals can be very toxic, but they are also essential and naturally occurring. Due to the fact that metals are essential and naturally occurring, a relatively low rank was assigned.

Table 10 Weighting factors for effects based on vulnerability of the endpoint to the different stressors, WE

Endpoint	Pharmaceuticals	Pesticides	Nutrients	Metals	Organic pollutants
Salmon and trout	6	3	1	2	5

When the ranking and weighting factors have been decided on, the relative risk can be calculated using the equations presented below. Equation 2 below represents the risk of adverse ecological effects in subarea *i* for the habitat *h*. This risk is obtained by summing up the risks posed by all of the stressors, *j*, which are emitted in one subarea.

$$\text{Risk\_subarea}_{ih} = \sum_{j} Risk\_subarea_{ijh} \qquad \text{(Eq. 2)}$$

[i] Lars Molander, fishing adviser at Länsstyrelsen Västra Götaland, personal meeting on the 8th of April 2009.

The risk of each stressor within a subarea is obtained by multiplying the relative exposure with the relative effect, Equation 2 below.

$$\text{Risk\_Subarea}_{ijh} = \text{Exposure}_{ijh} \times \text{Effect}_{jh} \quad \text{(Eq. 3)}$$

The exposure is derived from three different parameters, Equation 3. These are the ranking of the density of stressors (RS), the weighting factor that accounts for the relative amount each source emits of the different stressors (WS) and the ranking of density of habitats (RH). By multiplying RS with WS, a dimensionless number equivalent to a concentration is obtained. This is then multiplied with RH to account for habitats that can be exposed. This is done for all stressor sources, *k*.

$$\text{Exposure}_{ijh} = \sum_{k} \left(RS_{ki} \times WS_{jk} \right) \times RH_{hi} \quad \text{(Eq. 4)}$$

The effect is characterised by one factor, the vulnerability of the endpoint to the stressors, WE.

$$\text{Effect}_{jh} = WE_{jk} \quad \text{(Eq. 5)}$$

The final equation used to calculate the relative risk can be written as following:

$$\text{Risk\_subarea}_{ih} = \sum_{j} \left(\left(\sum_{k} \left(RS_{ki} \times WS_{jk} \right) \right) \times RH_{hi} \times WE_{jk} \right) \quad \text{(Eq. 6)}$$

The final results generated by the RRM are relative risk ranks for each individual stressor in each subarea. By summing up all the risks for the stressors in a subarea, the total relative risk of the subarea is obtained, as shown in Equation 6. The results from these calculations can be found in Appendix G.

Based on the relative risk scores, the different subareas can be divided into risk groups. These risk groups indicate the potential risk of adverse effects to the salmon and trout populations, based on density of stressor sources and habitats as well as the vulnerability of the endpoint to the emitted stressors. The subareas were divided into four risk groups according to the following criteria:

- Relative risk score = 0 → No risk
- 1 < Relative risk score < 100 → Low risk
- 101 < Relative risk score < 200 → Moderate risk
- 201 < Relative risk score → High risk

Risk characterisation using PEC/PNEC ratios

The PETAR procedure does not state how the risk characterisation with regard to stressors should be performed. The risk characterisation for the stressors was therefore based on the recommendations in the TGD on Risk Assessment (European Chemicals Bureau, 2003). According to the TGD, the first step of the risk characterisation is to calculate the PEC/PNEC ratios. For all PEC/PNEC ratios that are larger than one, there is a risk of adverse ecological effects since the environmental concentrations then are above the predicted no effect

concentration. Based on the obtained PEC/PNEC ratios, three different main conclusions can be drawn (European Chemical Bureau, 2003):

- Conclusion (i): There is need for further information or testing;
- Conclusion (ii): There is at present no need for further information and/or testing and no need for risk reduction measures beyond those which are being applied already;
- Conclusion (iii): There is need for limiting the risks; risk reduction measures, which are already being applied, shall be taken into account.

If all available information have been used to derive the PEC and PNEC values and if these ratios are found to be less than or equal to one for each of the compartments, conclusion (ii) should be applied. If any of the PEC/PNEC ratios for the different environmental matrices are greater than one, it is necessary to judge if further information or testing can lower the PEC/PNEC ratio. If that is possible, then further studies should be performed in order to refine the ratio. This corresponds to conclusion (i). If the refined PEC and PNEC values still generate a ratio greater than one, or if the first calculated ratio could not be lowered by further studies, conclusion (iii) should be drawn, that is there is a need for risk reduction.

5.1.4 Uncertainty and sensitivity analyses

Besides choosing methods for how to perform the analysis, it is also important to consider how to handle data quality issues and uncertainties. In the TGD (2003), there are some recommendations on data quality in risk assessments and how to handle uncertainties within the datasets when assessing the risk posed by the stressors. It is central to be aware of the background information of the data sets in order to know what conditions the data can be applied to, what the limitations are, as well as how much, and what kind of uncertainty that is connected to the data sets. If this information is not known, the transparency of the risk assessment and subsequently also the scientific value of the assessment are lost.

The following assessments were made in order to assess the uncertainty of the input to the risk assessment. Before using measured data in the exposure assessment, the reliability of the data needs to be assessed. To assess the reliability of the data obtained from literature, the sampling and analysis strategies for the different studies were reviewed. Based on these reviews, a qualitative assessment of the reliability was made. After that, the representativeness of the data compared to the emission situation must also be evaluated. Comparing the conditions at the locations of the studies with the conditions at Göta älv performed this. Where there were large differences, it could be of interest to recalculate the literature data in order to better match the conditions in Göta älv. It is also necessary to evaluate the data that are used for the effect characterisation, such as ecotoxicity data. The ecotoxicity data were generally used to determine PNECs for the different environmental matrices. The reliability of the PNECs was assessed by looking at the data upon which they were derived as well as the method used. This information can be found in Appendix F.

The sensitivity of the relative risk model was also assessed since it was identified as having potentially high uncertainty. Before the sensitivity analysis could be performed, the most sensitive parameters first had to be identified. After having done so, a deterministic approach was used to estimate the sensitivity (Morgan and Henrion, 1992). When using a deterministic approach, one parameter is changed while the others are kept constant at the original values. This procedure was performed three times in order to assess the sensitivity of the three parameters that were identified as having the largest effect on the output of the model.

5.2 Results

5.2.1 Problem formulation

The purpose of the problem formulation step is to select an appropriate assessment endpoint as well as to create a conceptual model and an analysis plan. In the sections below, the assessment endpoint and the conceptual model for the regional risk assessment are presented. The analysis plan is presented in section 5.1.1 above, the method section, since it describes how the analysis was performed.

Assessment endpoints

In the planning phase the aquatic ecosystems in Göta Älv were identified to be at highest risk for exposure to the selected stressors. The County Administrative Board has identified salmon and trout as species that are highly desirable to protect and preserve in the ecosystem (Göta älvs vattenvårdsförbund, 2006). One reason for this is that Göta älv was previously a river with large and viable salmon and trout populations that was highly valued by the sport fishing community. As the hydro electrical power plants were built in the river, the populations decreased and the native salmon fish species in the mainstream of Göta älv was extinct. Though, in the tributary flow of Grönån, experts still believe that there is a possibility that one of the native salmon species still exists (Jacobsen and Johansson, 1999). Consequently, it has been given a high protective value. To maintain Göta älv as a good fishing ground for salmon fish, salmon from the nearby stream Säveån is implanted into Göta älv each year. Fish communities are often good indicators of health of aquatic ecosystems since they are susceptible to many different stressors, their life span is long in comparison to other species in the ecosystem and they respond to effects on other compartments in the ecosystem (Moraes, 2002). Besides direct exposure to stressors in the water or sediments, fishes can also be exposed via their food. Thus, the salmon population fulfils all requirements for an appropriate assessment endpoint that are stated in section 5.1.1; it is included in management goals, it is ecologically relevant as well as being susceptible to the considered stressors.

The salmon, *Salmo salar*, which exists in the river Göta älv, belongs to the group of anadromous fish, which are migratory fishes (Jacobsen and Johansson, 1999). These fishes reproduce in fresh water and the young fishes, called smolt, usually stay there for one or two years after being hatched. The salmon then usually migrate to the northern parts of the Atlantic. After having spent one or more years in the Atlantic, the salmon returns to the rivers at the Swedish west coast to reproduce. The salmon usually reproduce in the larger tributaries of Göta älv. The majority of the fishes die after having reproduced once.

The trout, *Salmo trutta*, also belongs to the group of anadromous fish. However, studies of the Göta älv trout populations have shown that there are also stationary populations of trout that do not migrate and instead live their entire lives in Göta älv and its tributaries (Jacobsen and Johansson, 1999). In contrast to the salmon, the trout reproduce in smaller streams and brooks, which are generally located higher up in the tributaries. A problem for the trout population is the migratory obstacles that humans have built in the tributaries, which prevent the fish from reaching their playgrounds. To maintain the trout populations in the river and tributaries, work is done in order to remove migratory obstacles according to Molander[i]. Both salmon and trout reproduce during October to December.

[i] Lars Molander, fishing adviser at Länsstyrelsen Västra Götaland, personal meeting on the 8th of April 2009.

Studies have shown that the early life stages of the salmon fishes, smolt and juvenile fish, are extra sensitive to toxic exposure (Di Giulio and Hinton, 2008). These life stages are therefore of interest to study regarding salmon since they spend these life stages in Göta älv. Since the salmon and trout are predatory fishes, they can get exposed to stressors that biomagnify.

Conceptual model

The conceptual model that was developed for this part of the assessment describes which stressors the different sources emit and to which environmental matrices they are emitted (see **Figure 6**). It also describes how the stressors can move between the different environmental matrices and where they finally will end up.

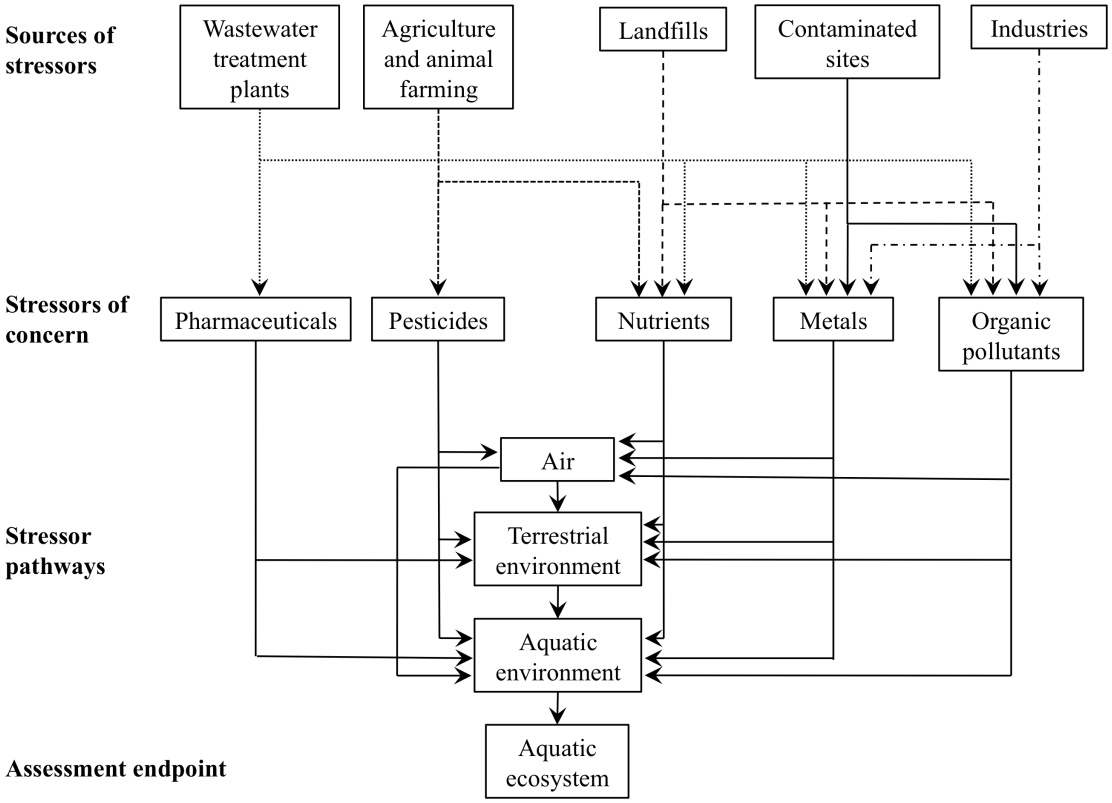


Figure 6 Simplified conceptual model linking sources of stressors to assessment endpoints

In the figure, it can be seen that the stressors are first emitted to several different environmental matrices, but that they can end up in the aquatic environment. This conceptual model was used as the basis for the calculations of predicted environmental concentrations (PEC); the information from the conceptual model was used in order to determine which sources to include in the calculations.

5.2.2 Analysis phase

The results from the analysis phase constitute the foundation for the risk characterisation. In this section, all necessary data to perform the risk characterisation will be presented.

Exposure characterisation

The first part of the results from the analysis phase consists of the exposure characterisation. In the following sections, the geographical distribution of stressor sources and habitats as well as predicted environmental concentrations of the stressors will be presented.

Geographical distribution of stressor sources

In order to analyse the geographical distribution of stressor sources, the study area was divided into different subareas based on hydrological data. Each subarea represents a smaller drainage area within the drainage area of Göta älv. In **Figure 7** below, there is a map over the study area and the different subareas. The subareas are numbered according to the order in which they drain to the river, in the direction from Vänersborg to Göteborg.

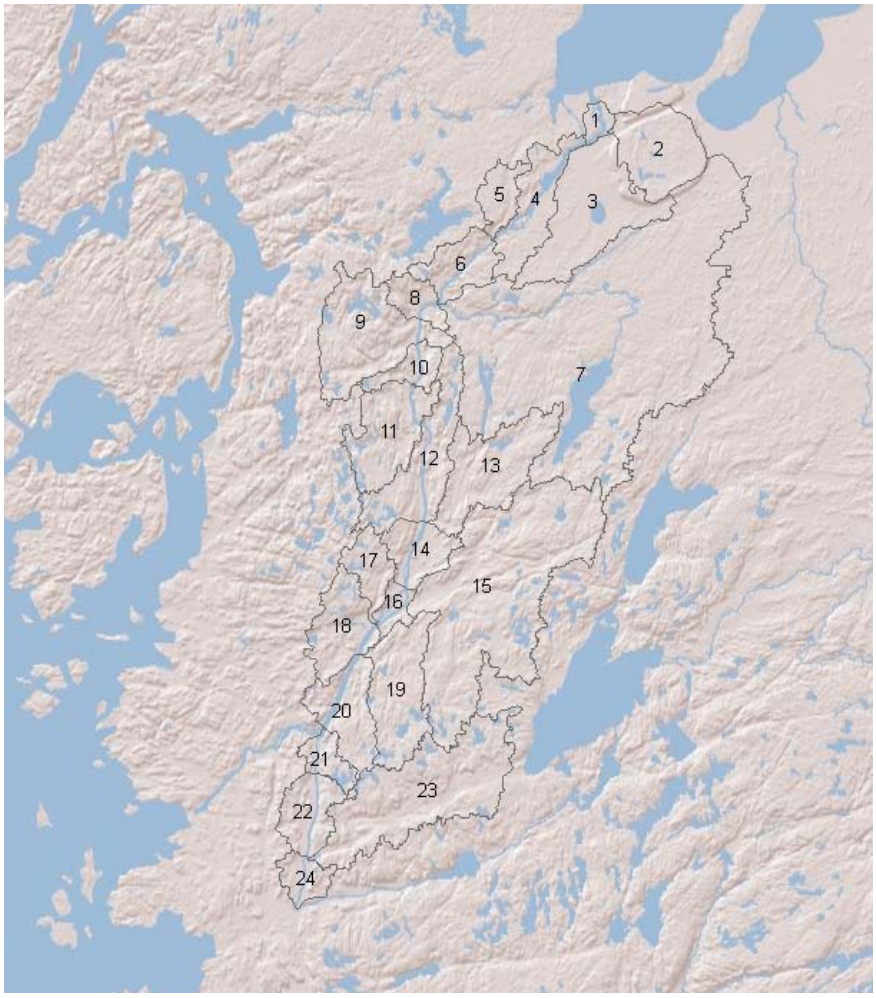
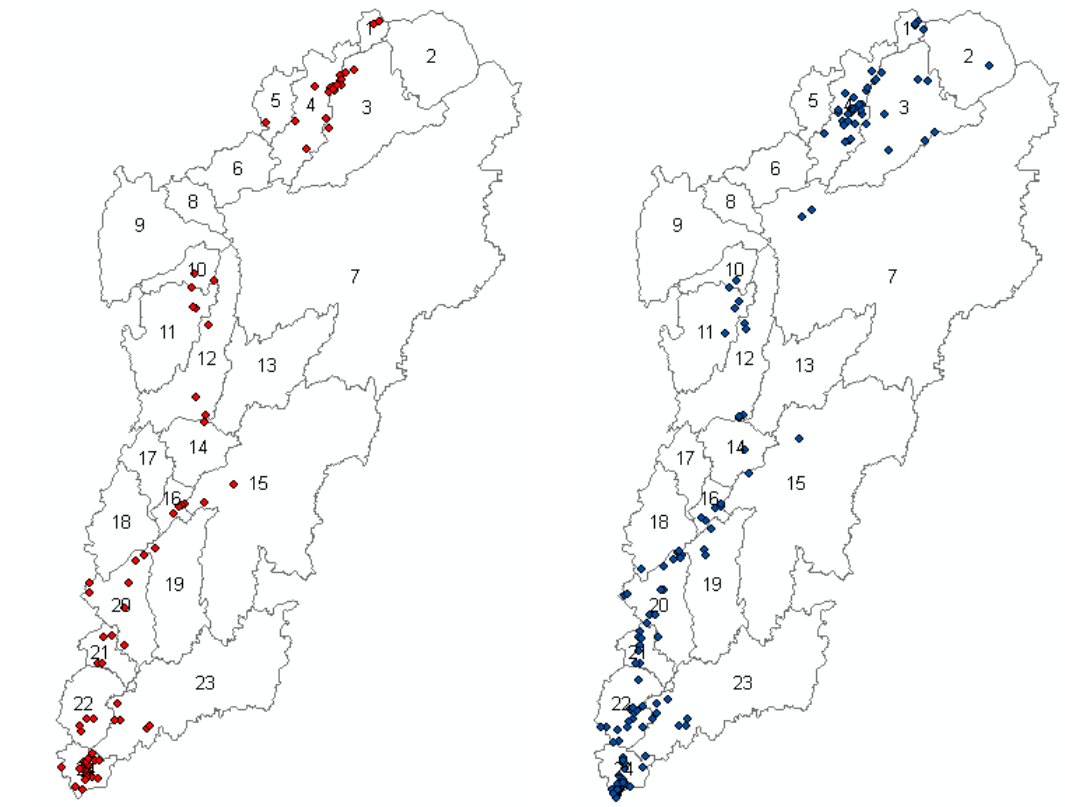


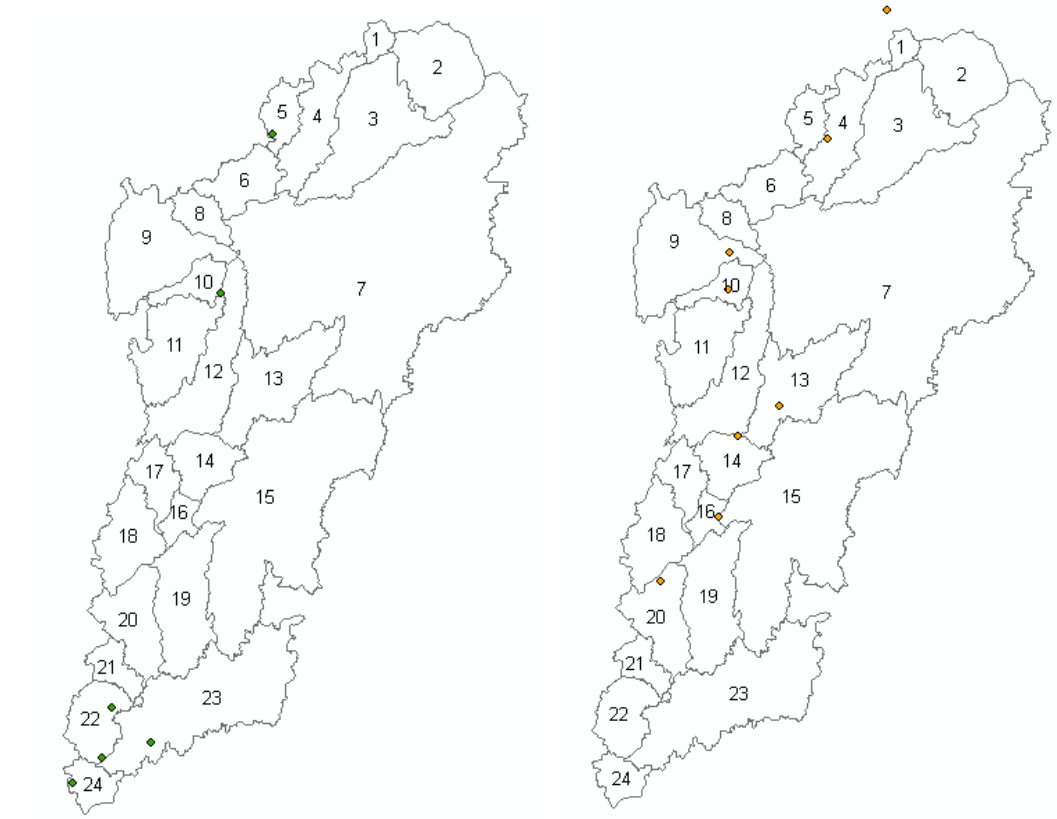
Figure 7 Map over the study area with all of the subareas marked and numbered

In **Figure 8** and **Figure 9** below, the distribution of the stressor sources can be seen. **Figure 8 a and b** shows the geographical distribution of the industries and contaminated sites located within the study area while **Figure 9 a and b** shows the distribution of landfills and wastewater treatment plants. As can be seen in **Figure 11 b**, there is one wastewater treatment plant that is located outside of the study area. This WWTP is located in Vänersborg in one of the catchment areas that drain directly to Lake Vänern, but since this WWTP releases its effluent to Göta älv it was included. The locations of some of the landfills are not exact due to lack of coordinates for the sites. Instead, coordinates for the landfills were approximated using old maps. Complete lists of all the stressor sources within the study area, with information about each source, can be found in Appendices A to D.



a. **b.**

Figure 8 Maps over the study area showing the distribution of stressor sources: a.) Industries, b.) Contaminated sites



a. **b.**

Figure 9 Maps over the study area showing the distribution of stressor sources: a.) Landfills, b.) Wastewater treatment plants

In **Table 11** below, the amount of agricultural land in the different subareas is shown, both given as square kilometres and as percentage of the total area of the subarea.

Table 11 Distribution of agricultural land in the different catchment areas

Catchment area	Total catchment area (km^2)	Agricultural land (km^2)	Agricultural land (%)
1	7,12	0,84	11,8
2	48,66	2,21	4,6
3	78,53	45,02	57,3
4	40,55	4,89	12,1
5	17,09	2,65	15,5
6	28,26	6,54	23,1
7	395,57	96,76	24,5
8	15,75	4,74	30,1
9	69,7	6,76	9,7
10	15,64	3,80	24,3
11	48,95	7,21	14,7
12	57,04	8,34	14,6
13	60,42	15,44	25,6
14	26,64	10,44	39,2
15	197,49	36,18	18,3
16	10,2	4,16	40,8
17	20,18	1,35	6,7
18	37,66	9,23	24,5
19	55,6	11,40	20,5
20	38,57	4,21	10,9
21	14,52	0,99	6,8
22	29,62	2,11	7,1
23	112,09	20,53	18,3
24	15,09	0,04	0,2
Total	1440,94	305,85	21,2

Source: SMHI, 2008.

In **Table 12** below, a summary of where the different stressors sources are located is presented. As can be seen in the table, there are some subareas with a higher number of stressor sources than others (marked in bold text). These are potential high-risk areas.

Table 12 Number of stressor sources within the different subareas

Catchment area	WWTP	Contaminated sites	Industries	Landfills[1]	Agricultural land (%)
1	0	5	2	0	11,8
2	0	1	0	0	4,6
3	0	10	11	0	57,3
4	**Arvidstorp WWTP**	**26**	**6**	**0**	**12,1**
5	0	0	1	1	15,5
6	0	0	0	0	23,1
7	0	2	0	0	24,5
8	0	0	0	0	30,1
9	Hjärtum WWTP	0	0	0	9,7
10	0	1	1	0	24,3
11	Ellbo WWTP	4	3	0	14,7
12	0	6	4	1	14,6
13	Nygård WWTP	0	0	0	25,6
14	Lödöse WWTP	1	1	0	39,2
15	0	2	3	0	18,3
16	Älvängen WWTP	5	3	0	40,8
17	0	0	0	0	6,7
18	Diseröd WWTP	1	0	0	24,5
19	0	3	1	0	20,5
20	**0**	**15**	**7**	**0**	**10,9**
21	0	7	4	0	6,8
22	**0**	**15**	**6**	**2**	**7,1**
23	0	8	4	1	18,3
24	**0**	**30**	**20**	**1**	**0,2**
Total	7	142	77	6	21,2

[1] The dataset is only complete for the municipality of Göteborg. Data is missing regarding closed landfills for the other municipalities.

Geographical distribution of habitats

The geographical distribution of trout and salmon in the river basin of Göta älv is shown in **Figure 10** below. The data used here was obtained from Länsstyrelsen Västra Götaland via Molander[i]. The different habitats have been identified by samples fishing in the tributaries.

[i] Lars Molander, fishing adviser at Länsstyrelsen Västra Götaland, personal meeting on the 8th of April 2009.

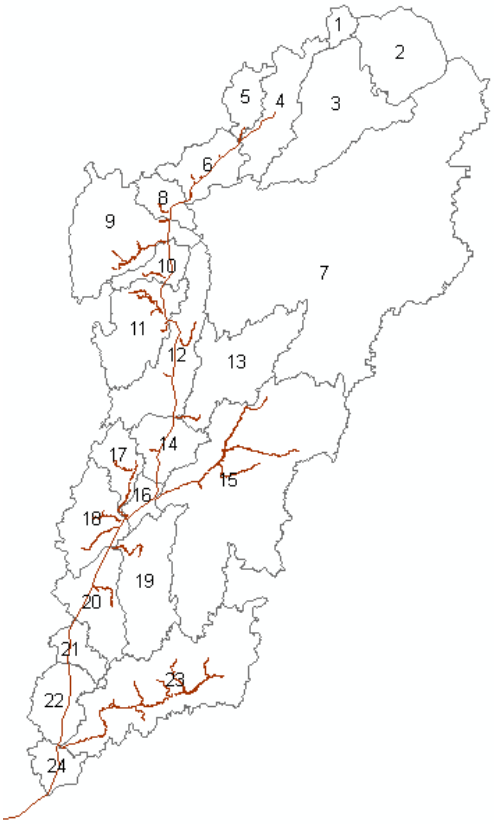


Figure 10 Map showing known habitats of salmon and trout within the study area

Some of the tributaries have been identified as being of more importance than others for the salmon and trout, due to the fact that they are important playgrounds. Molander[i] have identified the following tributaries as important playgrounds for salmon and trout:

- Sollumsån, subarea 9
- Brattorpsån, subarea 9
- Västerlandaån, subarea 11
- Ryrsjöbäcken, subarea 12
- Grönån, subarea 15
- Solbergsån, subarea 17
- Sköldsån/Hältorpsån, subarea 19
- Lärjeån, subarea 23

Predicted environmental concentrations – measured data

There is not site-specific data available for all stressors; therefore the data used here is a mixture of site specific and not site-specific data. For the stressors where site-specific data was not available, data have been gathered from Swedish and Danish measurement campaigns published in scientific articles. Some of these data have been recalculated based on use of the stressor to better fit the conditions in Göta älv.

Among the stressors, measurements of metals, nutrients and PAH have been performed in the Göta älv, e.g. in studies by Länsstyrelsen, Göteborgs Hamn and Göta älvs vattenvårdsförbund

[i] Lars Molander, fishing adviser at Länsstyrelsen Västra Götaland, personal meeting on the 8th of April 2009.

as well as by the Swedish University of Agricultural Sciences (SLU) (2009). For the soluble fraction of PAH, the measurements were performed at Ringön just outside the study area in the harbour of Göteborg (Göta älvs vattenvårdsförbund, 2007). This data is not optimal since there are many sources of PAHs in the harbour of Gothenburg, but due to limited site-specific data, it is still used here. Concentrations of metals and nutrients in the river water are continuously measured by SLU in a monitoring programme of the water chemistry in Swedish lakes and watercourses. Here, data from the two stations Vargön in Vänersborg and Alelyckan north of Göteborg have been used. At these stations, water chemistry parameters are measured and recorded each month, and have so been since the 1980s for most parameters.

The data for pharmaceutical residues in surface water were compiled from five different studies (Bendz *et al*, 2005; Naturvårdsverket, 2008a; Andersson *et al*., 2005; Paxéus, 2004; Ferrari *et al*., 2003). All studies were performed in Sweden, with the aim to establish background concentrations, concentrations in surface waters, as well as concentrations in WWTP effluent. One study was performed at the stream Höje in Skåne, downstream a WWTP (Bendz *et al*., 2005). The second study was conducted in several Swedish recipients (Naturvårdsverket, 2008a). The third study, performed by Andersson *et al*. (2005), was a part of the Swedish national Screening Programme during the year of 2005, and samples where therefore taken at many different locations in Sweden. The two remaining studies, Paxéus (2004) and Ferrari *et al*. (2003) focused on measuring pharmaceutical residues in water to and from WWTPs.

For the pesticides, data was taken from a study performed in two rivers located in an agricultural area in Skåne (Adielsson and Kreuger, 2008). The study showed that there were differences in the use (kg/ha) of the two pesticides in Skåne and the Göta älv river valley; diflufenikan is used more in the Göta älv river valley than in Skåne while MCPA is used less. The dilution in these two rivers is lower than in Göta älv. As a result, environmental concentrations are probably lower in Göta älv.

Finally, for the two organic pollutants HBCD and PFOS, two screening studies performed by the Swedish Environmental Research Institute IVL, were used. For both these studies, the measurements were performed at different locations in Sweden, both at non-polluted and urban locations (Sternbeck *et al*., 2001; Woldegiorgis *et al*., 2005). For PFOS, results from a Danish study have also been used (Bossi *et al*., 2008). This study focused on measuring the concentrations of PFOS in recipients close to WWTPs, landfills and industries.

In **Table 13** below, the measured concentrations of stressors from the different studies are compiled. There are relatively large ranges of concentrations for some stressors. This is most likely due to uncertainties and differences between background and urban concentrations. However, since the Göta älv river basin is highly industrialised and relatively urbanised, urban concentrations can be expected in the river. For the data measured in WWTP effluent, a dilution factor of 200 can be applied to get an approximation of the concentration in Göta älv. The dilution factor of 200 corresponds to the dilution factor of wastewater in Göta älv at low flow (Göta älvs vattenvårdsförbund, 2006; Åström and Pettersson, 2007). For calculations of dilution factors for WWTP effluent in Göta älv, see Appendix C.

Table 13 Predicted environmental concentrations (PEC) of stressors in water and biota obtained from literature

Stressors	Conc. WWTP effluent (µg/l)	PEC water (µg/l)	PEC biota (µg/kg ww)	References
Pharmaceuticals				
Diclofenac		0.0005-0.12		Naturvårdsverket, 2008a; Bendz *et al*., 2005; Andersson *et al*., 2005; Paxéus, 2004
Propranolol		0.00015-0.01		Naturvårdsverket, 2008a; Bendz *et al*., 2005
Carbamazepine		0.1-0.5		Bendz *et al*., 2005; Ferrari *et al*., 2003
Ethinylestradiol		0.00005-0.001		Naturvårdsverket, 2008a, Andersson *et al*., 2005
Pesticides				
MCPA		0.17		Adielsson and Kreuger, 2008
Diflufenican		0.0055		Adielsson and Kreuger, 2008
Metals				
Cu		1-3		SLU, 2009a; SLU, 2009b
Zn		2-14		SLU, 2009a; SLU, 2009b; Naturvårdsverket, 2000
Cr		0.1-2.4		SLU, 2009a; SLU, 2009b
Hg		0.0005-0.022		SLU, 2009a; SLU, 2009b; Johansson and Skrapste, 2003; Naturvårdsverket, 2000
Organic pollutants				
PAH		0.000013		Göta älvs vattenvårdsförbund, 2007; Johansson and Skrapste, 2003
HBCD	0.031			Sternbeck *et al*, 2001
PFOS		0.002-0.039	1.2-156[1]	Woldegiorgis *et al*., 2006; Bossi *et al.*, 2008
Nutrients				
Nitrogen (N-tot)		700		SLU, 2009a; SLU, 2009b
Phosphorous (P-tot)		10-20		SLU, 2009a; SLU, 2009b

[1] Concentration measured in fresh water fish

Predicted environmental concentrations (PECs) – model calculations

To quantify the risk that the stressors pose, the potential concentrations of the stressors in Göta älv were calculated. Since no measurements were performed within this study, concentrations were derived from previous measurements published in scientific articles. Before any of the data were used, their adequacy and reliability were assessed. For more detailed descriptions of the data sets, see Appendix C. For the pharmaceuticals and the pesticides, enough data were found to make model calculations of PECs in the river. However, for the organic pollutants, data was found for all relevant sources. The concentrations of PFOS are therefore only based on emissions from WWTPs while no concentrations could be calculated at all for HBCD.

The calculated PECs in the river can be seen in **Figure 11** to **Figure 17**. The three different lines in the graphs show the results from three different scenarios:

- High dilution
- Medium dilution
- Low dilution

In **Figure 11** to **Figure 14**, the modelled concentrations of the four pharmaceuticals are shown. As can be seen in the figures, the graphs in all figures have the same shape since they are emitted by the same source, namely the WWTPs. For these substances, the initial concentration is not equal to zero, which is due to the fact that the emissions from the WWTP in Vänersborg, called Holmängen, was included in the calculations. They also show an increasing trend from Vänersborg to Göteborg.

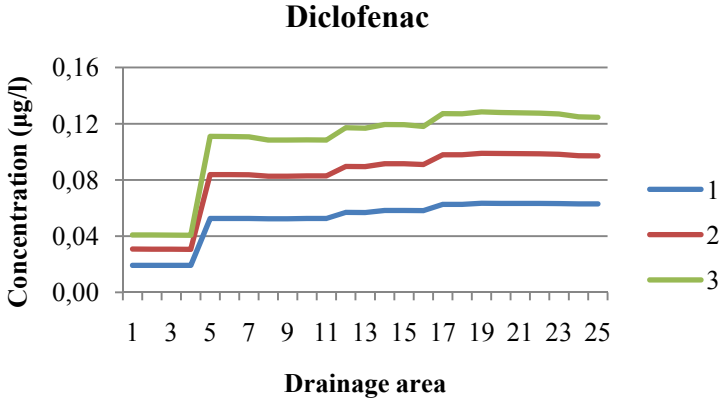


Figure 11 Accumulating concentrations of diclofenac obtained from model calculations

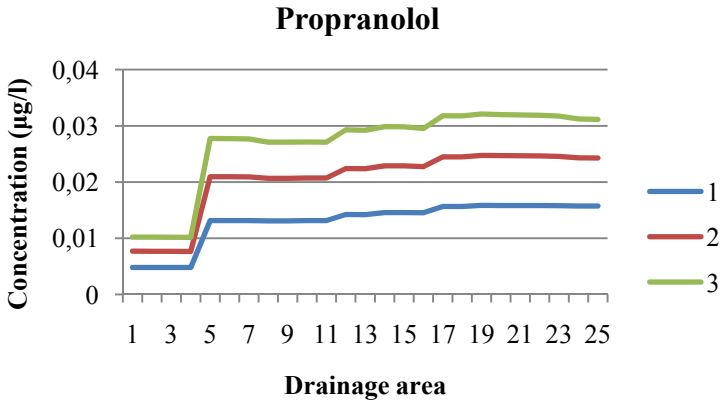


Figure 12 Accumulating concentrations of propranolol obtained from model calculations

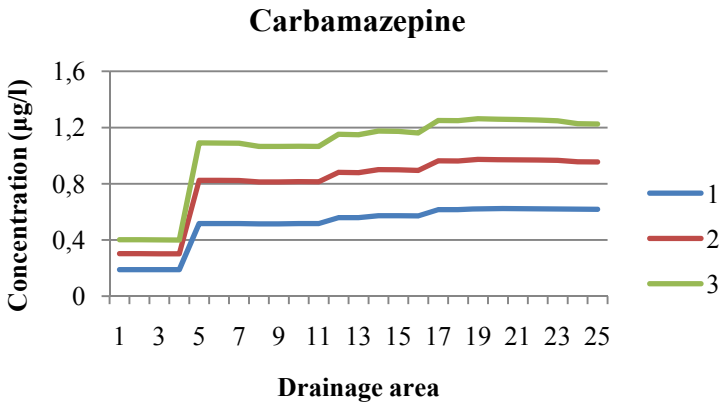


Figure 13 Accumulating concentrations of carbamazepine obtained from model calculations

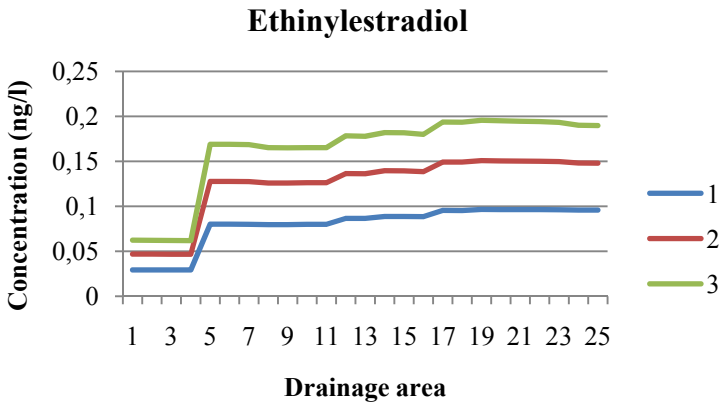


Figure 14 Accumulating concentrations of ethinylestradiol obtained from model calculations

Figure 15 and **Figure 16** below show how the concentrations of the two pesticides MCPA and diflufenican change from Vänersborg to Göteborg.

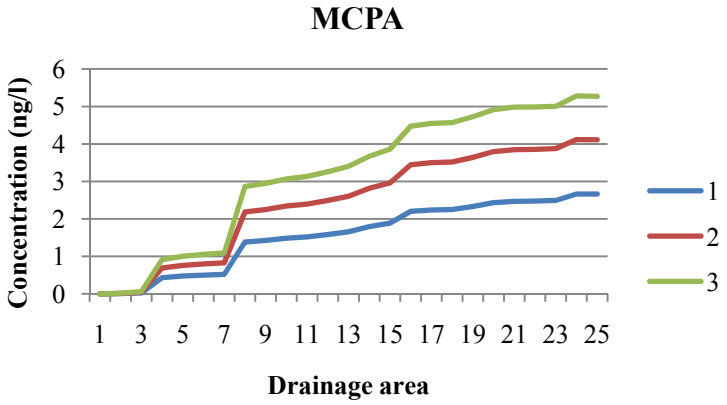


Figure 15 Accumulating concentrations of MCPA obtained from model calculations

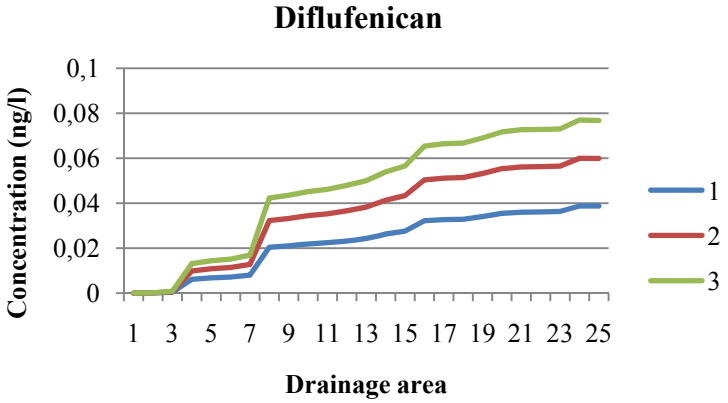


Figure 16 Accumulating concentrations of diflufenican obtained from model calculations

Amongst the organic pollutants, PFOS was the only substance for which model calculations could be performed due to lack of data. The PEC for PFOS was calculated in the same way as the PECs for pharmaceuticals. As can be seen in **Figure 17**, the concentrations of PFOS change in the same way as the pharmaceuticals.

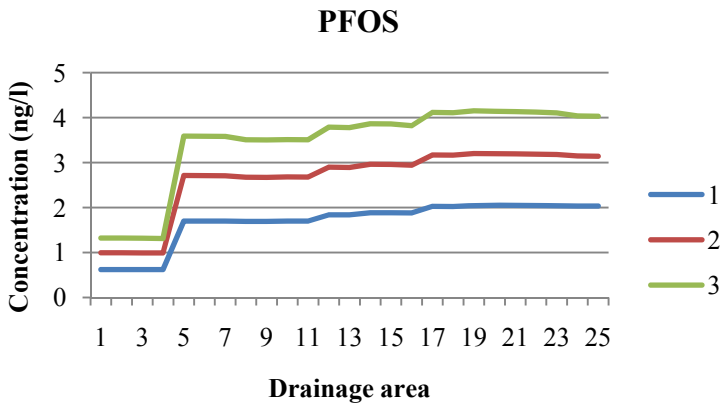


Figure 17 Accumulating concentrations of PFOS obtained from model calculations

Based on the model calculations, ranges of PECs for seven of the studied stressors were obtained. In **Table 14** below, the lowest concentrations at maximum dilution and the highest concentrations at minimum dilution of the different stressors are shown.

Table 14 Predicted environmental concentrations (PEC) for stressors in Göta älv surface water obtained from model calculations

Stressors	PEC water	References
Pharmaceuticals		
Diclofenac (µg/l)	0.019-0.12	Bendz *et al*., 2005
Propranolol (µg/l)	0.0048-0.031	Bendz *et al*., 2005
Carbamazepine (µg/l)	0.18-1.23	Bendz *et al*., 2005
Ethinylestradiol (ng/l)	0.029-0.19	Adolfsson-Erici[i]
Pesticides		
MCPA (ng/l)	0.008-5.27	Kreuger *et al*., 2003; Kreuger *et al*., 2004; Törnqvist *et al*., 2005; Adielsson *et al*., 2006
Diflufenican (ng/l)	0.0001-0.077	Kreuger *et al*., 2003; Kreuger *et al*., 2004; Törnqvist *et al*., 2005; Adielsson *et al*., 2006
Organic pollutants		
PFOS (ng/l)	0.62-4.03	Bossi *et al.*, 2008

In **Table 15** below, both PECs from literature and model calculations are shown. For the pharmaceuticals, both literature and model calculations show similar results. The pesticides and PFOS on the other hand show differences between literature and modelled data. The literature concentrations for these three stressors are higher than the modelled concentrations.

Table 15 Comparison between literature PECs and modelled PECs in water

Stressors	PEC water, literature (µg/l)	PEC water, model calculations (µg/l)
Pharmaceuticals		
Diclofenac	0.0005-0.12	0.019-0.12
Propranolol	0.00015-0.01	0.0048-0.031
Carbamazepine	0.1-0.5	0.18-1.23
Ethinylestradiol	0.00005-0.001	0.029-0.19
Pesticides		
MCPA	0.063[1]	0.000008-0.00523
Diflufenican	0.01[1]	0.0000001-0.000077
Metals		
Cu	1-3	
Zn	2-14	
Cr	0.1-2.4	
Hg	0.0005-0.022	
Organic pollutants		
PAH	0.000013	
HBCD	0.000016-0.00016[3]	
PFOS	0.002-0.039	0.00062-0.0040
Nutrients		
Nitrogen (N-tot)	700	
Phosphorous (P-tot)	10-20	

[i] Margaretha Adolfsson-Erici, Department of Applied Environmental Science, Stockholm University. E-mail conversation on the 17th of April 2009.

Effects characterisation

The second part of the results from the analysis phase consists of the effects characterisation. Which effects that the different stressors can cause have already been presented in section 4.2.3 above. Here, predicted no effect concentrations for the different stressors in water and biota will be presented.

Predicted no effect concentrations, PNEC

Below in **Table 16**, all of the PNEC data and their respective references can be seen.

Table 16 Predicted no effect concentrations (PNEC) in water and sediments for the selected stressors

Stressors	PNEC water (µg/l)	PNEC biota (µg/kg ww)	References
Pharmaceuticals			
Diclofenac	0.1		Naturvårdsverket, 2008a
Propranolol	0.005		Naturvårdsverket, 2008a
Carbamazepine	0.42		Ferrari *et al*., 2003
Ethinylestradiol	0.0001		Naturvårdsverket, 2008a
Pesticides			
MCPA	1.1		Naturvårdsverket, 2008c
Diflufenican	0.005		Naturvårdsverket, 2008c
Metals			
Cu	4		Naturvårdsverket, 2008c
Zn	8[1]		Naturvårdsverket, 2008c
Cr	3		Naturvårdsverket, 2008c
Hg	0.1		Naturvårdsverket, 1999
Organic pollutants			
PAH	0.2		Naturvårdsverket, 1999
HBCD	0.03		Naturvårdsverket, 2008c
PFOS	3	6	Naturvårdsverket, 2008c

[1] The alkalinity in Göta älv is approximately 30 mg $CaCO_3$/l (Göta älvs vattenvårdsförbund, 2007). This PNEC should be compared to the anthropogenic fraction of Zn, not including the background concentration.

Based on the Swedish Environmental Quality Criterion for lakes and watercourses (Naturvårdsverket, 2000), an assessment of the nutrient status in Göta älv can be made, see **Table 17** below. For total nitrogen, the concentrations of total nitrogen in the river ranged between 500-1,200 µg/l. Compared to the environmental quality criteria, this corresponds to a nitrogen concentration that is moderately high to high. For total phosphorous, the measurements showed that the concentration ranges between 10 and 20 µg/l, which correspond to a state that is nutrient-poor to moderately nutrient rich. Thus Göta älv cannot be classified as eutrophic.

Table 17 Quality criteria for assessment of eutrophication in lakes

Stressors	Quality criteria (µg/l)	References
Nutrients		
- Nitrogen (N-tot)	Low concentrations ≤ 300 Moderately high concentrations 300-625 High concentrations 625-1250	Naturvårdsverket, 2000
- Phosphorous (P-tot)	Low concentrations ≤ 12.5 Moderately high concentrations 12.5-25 High concentrations 25-50	Naturvårdsverket, 2000

5.2.3 Risk characterisation

In the following two sections, the results from the two different risk characterisations are presented.

The relative risk model (RRM)

When using the RRM a ranking of subareas were obtained. The results are only relative and indicate which subareas that are of higher risk than others. The results of the ranking are shown in **Table 18** and **Table 19** below.

Table 18 Ranking of stressor sources for the different subareas

	Subareas																							
Sources	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**	**11**	**12**	**13**	**14**	**15**	**16**	**17**	**18**	**19**	**20**	**21**	**22**	**23**	**24**
WWTP	0	0	0	6	0	0	0	0	2	0	2	0	2	2	0	4	0	2	0	0	0	0	0	0
Contaminated sites	4	2	2	4	0	0	2	0	0	2	2	2	0	2	2	4	0	2	2	2	4	4	2	6
Industries	4	0	2	2	2	0	0	0	0	2	0	2	2	4	0	0	2	2	4	2	4	4	2	6
Landfills	0	0	0	0	4	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	6	2	6
Agriculture	2	2	6	2	2	4	4	4	2	4	2	2	4	4	2	4	2	3	2	2	2	2	2	0

Table 19 Ranking of habitats for the different subareas

	Subareas																							
Habitats	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**	**11**	**12**	**13**	**14**	**15**	**16**	**17**	**18**	**19**	**20**	**21**	**22**	**23**	**24**
Salmon and trout habitats	0	0	0	1	1	1	0	1	2	1	2	2	1	1	2	1	2	1	2	1	1	1	2	1

The relative risk to the salmon and trout populations within the different subareas is presented in **Figure 18** below. The different colours of the bars in the chart indicate which and how much the different stressors contribute to the risk in each subarea.

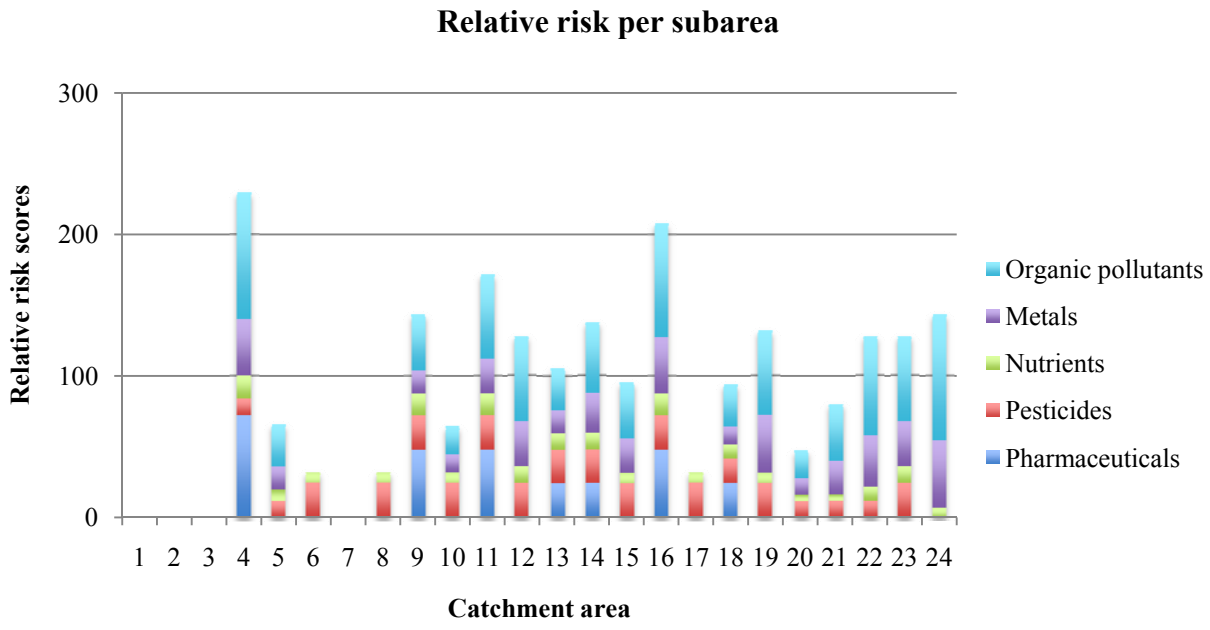


Figure 18 Relative risk per subarea to the salmon and trout populations

In **Figure 19** below the relative risk of the different subareas is presented. In the map, the different shades of grey show which risk group the different areas have been ranked into; white represents no relative risk, light grey low relative risk, medium grey moderate relative risk and dark grey high relative risk.

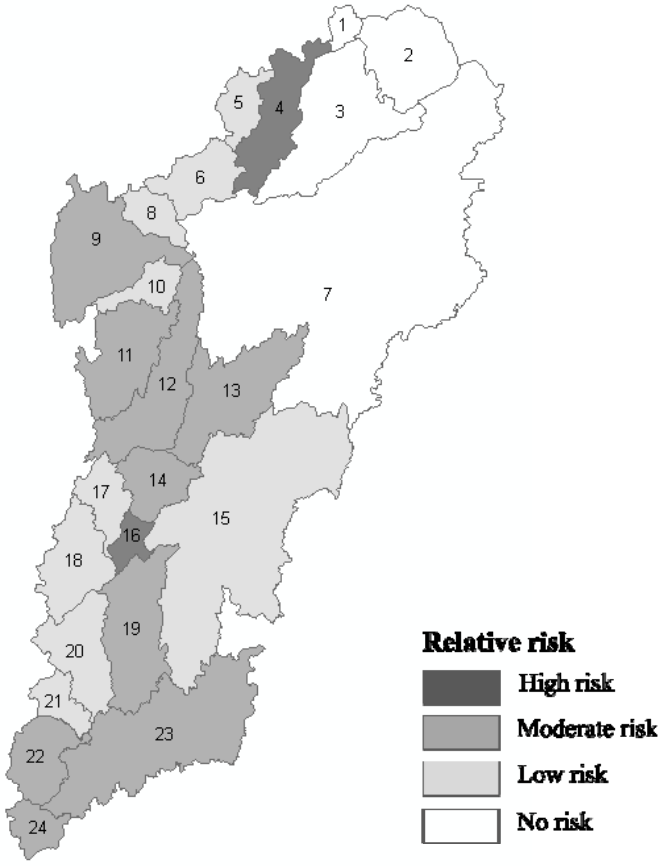


Figure 19 Map over the study area showing the relative risk ranks for the different subareas

According to the risk group criteria presented above, two areas are considered as high relative risk areas. Those are subareas 4 and 16. Subarea 4 is located at Trollhättan. For this area, the density of stressor sources is high since there is a large industrial area located within the subarea (the Stallbacka industrial area). The largest WWTP in the study area is also located within this subarea, contributing to the risk of the area. The two groups of stressors that contribute the most to the risk in this subarea are pharmaceuticals and organic pollutants. The other subarea that is classified as being of high risk is number 16, which covers the area of the town Älvängen in Ale. As for subarea 4, there is a WWTP located within subarea 16, although this WWTP has less person equivalents connected. The majority of the subareas are classified as either moderate or low relative risk areas. There are four areas that have been classified as no risk areas. This is because there are no registered salmon and trout habitats within these areas.

Risk characterisation using PEC/PNEC ratios

In **Table 20** below, calculated PEC/PNEC ratios for the different stressors in the relevant environmental matrices are presented. As can be seen in the table, the ratios are presented as ranges. The ranges represent the minimum and maximum concentrations that have been reported in the scientific literature. As can be seen in **Table 20**, there are several stressors for which the upper range of the PEC/PNEC ratios is above one.

Table 20 Calculated risk ratios based on PEC and PNEC values for each stressor

Stressors	PEC/PNEC water, model calculations	PEC/PNEC water, literature	PEC/PNEC biota, literature
Pharmaceuticals			
Diclofenac	0.19-1.2	0.0005-1.2	
Propranolol	0.96-6.2	0.03-2	
Carbamazepine	0.43-2.9	0.24-1.19	
Ethinylestradiol	0.29-1.9	0.5-10	
Pesticides			
MCPA	7.3×10^{-3}-0.0048	0.06	
Diflufenican	2×10^{-5}-0.015	2	
Metals			
Cu		0.25-0.75	
Zn		0.25-1.75	
Cr		0.033-0.8	
Hg		0.005-0.022	
Organic pollutants			
PAH		0.00007	
HBCD		0.0005-0.005	
PFOS	2.1×10^{-4}-0.0013	0.0007-0.013	0.2-26

In **Table 21**, conclusions regarding the risk and/or the need for more information on the different stressors have been drawn.

Table 21 Conclusions based on the risk characterisation

Stressors	Risk conclusions	Comments
Pharmaceuticals		
- Diclofenac	Conclusion (i)	PEC/PNEC >1for both model calculations and measured data
- Propranolol	Conclusion (i)	PEC/PNEC >1for both model calculations and measured data
- Carbamazepine	Conclusion (i)	PEC/PNEC >1for both model calculations and measured data
- Ethinylestradiol	Conclusion (i)	PEC/PNEC >1for both model calculations and measured data
Pesticides		
- MCPA	Conclusion (i)	PEC/PNEC >1for measured data
- Diflufenican	Conclusion (i)	PEC/PNEC >1for measured data
Metals		
- Cu	Conclusion (ii)	PEC/PNEC <1
- Zn	Conclusion (iii)	PEC/PNEC >1[1]
- Cr	Conclusion (ii)	PEC/PNEC <1
- Hg	Conclusion (ii)	PEC/PNEC <1
Organic pollutants		
- PAH	Conclusion (i)	PEC/PNEC >1for measured data, no model calculations performed due to lack of data
- HBCD	Conclusion (i)	No model calculations performed due to lack of data
- PFOS	Conclusion (i)	PEC/PNEC >1for measured data

[1] For the majority of the measurements, the PEC/PNEC ratio is below 1, but at single occasions the ratio is higher than 1.

As can be seen in the table, all conclusions according[i] to TGD have been drawn for the studied stressors. For the pesticides and the organic pollutants, conclusion (i) was drawn due

[i] Conclusion (i): There is need for further information or testing;
Conclusion (ii): There is at present no need for further information and/or testing and no need for risk reduction measures beyond those which are being applied already;
Conclusion (ii): There is at present no need for further information and/or testing and no need for risk reduction measures beyond those which are being applied already

to the large discrepancies between the model calculations and the measured data (see also **Table 20**). Testing within the study area may result in a lowering of the ratios and thereby the risk for these substances, since the highest ratios were obtained using data that is not site-specific.

The resulting PEC/PNEC ratios from both measured data and model calculations correspond well for the pharmaceuticals. Here, conclusion (i) is drawn since the generated ratios are larger than one. For these substances, further testing will probably confirm the predicted environmental concentrations presented here.

For the metals, there is a large, adequate and reliable dataset available that has been measured in Göta älv. Further testing will with high certainty not change the ratios calculated for the metals. Besides zinc, all metals have also obtained ratios smaller than one and conclusion (ii) is therefore drawn. For zinc, conclusion (iii) has been drawn since measure concentrations yielded ratios of above one.

5.2.4 Uncertainty and sensitivity analyses

In the sections below, the results from the uncertainty analysis and the sensitivity analysis for the different parts of the regional risk assessment will be presented.

The relative risk model (RRM)

Uncertainty analysis

In this study, the data for the input parameters, that is the distribution of stressor sources and the habitat distribution, have been obtained from thorough studies and can therefore be considered as having low uncertainty. The model structure on the other hand, how the ranks and weighting factors were designed, has been identified as having a higher degree of uncertainty.

Sensitivity analysis

Based on the uncertainty analysis, three parameters in the relative risk model were identified as having higher uncertainty than the others. These parameters are the stressor weighting factors, the effect weighting factors and the stressor ranking criteria. The sensitivity analysis was performed by quantifying the changes of the output as a result of changes in the input to the model. Three different sensitivity analyses were performed. In each analysis, only one parameter was changed. The input parameters were changed as following:

- Sensitivity analysis 1: Stressor weighting factors set as 0 or 1, indicating no release or release from the source without taking the emitted amounts into consideration.
- Sensitivity analysis 2: Effect weighting factors set as 1 for all stressors, thereby making the endpoint equally sensitive to all stressors.
- Sensitivity analysis 3: Stressor sources ranking criteria were set by calculating the highest density of each source within the entire study area and dividing the range into three equal parts.

The results from the different sensitivity analyses are shown in **Figure 20** below. In the figure, the results from the sensitivity analysis have been normalised to the original results of the relative risk ranking in order to facilitate comparison of the different results. The results were normalisation by dividing the scores for the subareas with the maximum score in each test. The relative distribution of the subareas in each test has not been altered when normalising the

results. As can be seen in the figure, the relative distribution of the risk ranks is quite stable. Even though the input parameters were changed, the final risk rank for most subareas was the same. For some subareas the risk rank changed, but most of the changes were caused by small variations in risk ranks for subareas that were close to one of the risk rank limits. For the subareas close to Göteborg, the changes caused by the input variations were larger. In sensitivity analysis 1, the relative risk increased a lot. This was because these areas contain many stressor sources that in the original risk rank were given low emission ranks. When not considering the amount of emitted stressors, the relative risk rank increased. Changing the effect weighting factors also generated some changes in the output results, although the changes were smaller than changes in stressor weighting factors. The sensitivity analyses show that the results from the relative risk model can be considered as relatively stable since there are no large changes in the ranking of the risk of the subareas, even when making large changes in the input parameters.

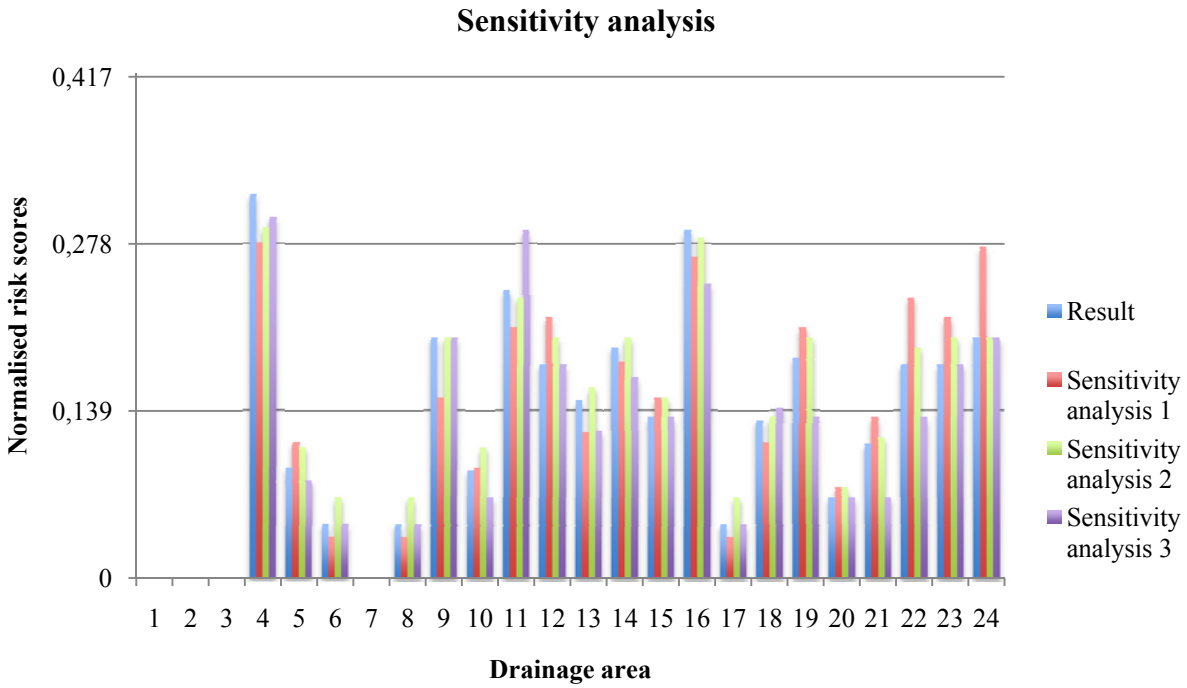


Figure 20 The original risk ranking results compared to the normalised results from the sensitivity analysis

PEC/PNEC ratios

Uncertainty analysis PEC – literature data

When analysing the uncertainties literature data, there are some uncertainties that appear as more common and important than others. These uncertainties are linked to spatial differences between the study area and the site of the measurements as well uncertainties due to sampling and analysis of results. As mentioned in previous sections, site-specific data are not available for all stressors. Due to differences in geography, hydrology, emission patterns of stressors etc, there will be differences in environmental concentrations between different sites. This is the single largest uncertainty in the literature data. The sampling and analysis strategies for the different studies can also affect the uncertainty of the results. In some studies, only a few samples has been taken, and the results of these samples will then only represent a snapshot of the reality, not accounting for variability. Results from such studies will inevitably contain

high uncertainties. The analysis method is also important. When the environmental concentrations are so low that they are close to the limit of detection, LOD, the analysis error can be higher than normal. Although, compared to the spatial uncertainties, the analysis and sampling uncertainties can be considered small.

In **Table 22** below, the uncertainties for the different datasets that have been used are presented. The uncertainties have been ranked with regard to the three most common types of uncertainties described above. The spatial uncertainties are ranked based on if the measurements have been performed in the study area or not. Site-specific measurements have been ranked as having no spatial uncertainties. Uncertainties linked to sampling were assessed based on the documented sampling strategy described in the different reports. The analysis uncertainties were ranked based on if the environmental concentrations are close to the limit of detection (LOD) for the instruments or not[i].

Table 22 Uncertainties in the used literature data

	Uncertainties			
Data set	**Spatial**	**Sampling**	**Analysis**	**Comment**
Pharmaceuticals:				
Bendz *et al.,* 2005	Yes	Low	High	Adequate, but not site specific
Andersson *et al.,* 2005	Yes	Low	High	Adequate, but not site specific
Pesticides:				
Adielsson and Kreuger, 2008	Yes	Low	High	Adequate and reliable, but not site specific
Metals and nutrients:				
SLU, 2009a	No	Low	Low	Adequate and reliable data
SLU, 2009b	No	Low	Low	Adequate and reliable data
Organic pollutants:				
Johansson and Skrapste, 2003	No	Low	High	No documented sampling and analysis strategy
Göta älvs vattenvårdsförbund, 2007	No	Low	High	Few samples, no documented analysis strategy
Sternbeck *et al*., 2001	Yes	High	High	Single sample
Woldegiorgis *et al.,* 2006	Yes	Low	High	Adequate, but not site specific
Bossi *et al.,* 2008	Yes	Low	High	Adequate, but not site specific

Uncertainty analysis PEC – model calculations

As for the relative risk characterisation, there are two main types of uncertainties, those connected to the model structure and those connected to the used data for the input parameters. As mentioned before, the model is a simple dilution model that calculates concentrations of stressors based on flow of the river and added amount of stressor. To account for differences in flow due to seasonal variations, concentrations have been calculated for three different scenarios. The uncertainty connected to the flow of water in which the stressors are diluted can therefore be considered as low.

The other part of the model consists of the calculations of the amounts of stressor that is added to the river. Here, there are some factors that can have an effect on the results of the calculations. A large fraction of the input data is not site specific. To account for this, the data has been recalculated to fit the conditions in the study area. Emissions from WWTPs, that is

[i] The analysis uncertainties where judged based on the uncertainty assessments for the different measurements as stated in the reports.

pharmaceuticals and PFOS, have been scaled to connected person equivalents while pesticides have been scaled according to use, average leakage and area of arable land. Even so, there are uncertainties within these data. Another source of uncertainties is that all relevant sources could not be included in the calculations due to lack of data. As a result, the concentrations of the stressors are underestimated. On the other hand, degradation processes have not been considered. Since degradation processes are not included, the concentrations are at the same time overestimated. Since these uncertainties affect the results in opposite directions, there is no clear indication that the calculated concentrations are over- or underestimated.

Uncertainty analysis PNEC derivation

Uncertainties in the PNEC values mainly arise from how the toxicity data were obtained, the amount of available toxicity data and the method used to derive the PNEC from the toxicity data. It will always be uncertainties in PNECs since they only are predicted no effect concentrations. For all pharmaceuticals but carbamazepine, the uncertainties connected to the PNECs are not known since there is no information on how they were derived. The PNEC for carbamazepine on the other hand can be considered as having relatively low uncertainty since it has been derived based on extensive amount of data. The uncertainties in the PNECs for the pesticides can be considered as moderate since they are based on a limited amount of data. Among the metals, the uncertainty of the PNECs for copper and zinc are low, while chromium has moderate uncertainties and mercury has high. For the last group of stressors, the uncertainties can be considered moderate to high based on the data upon which they are based. It is difficult to quantitatively assess the uncertainty of the PNEC values since all information on how they were developed are not known. More detailed information about which data the PNECs were derived upon and by what method, see Appendix F.

5.2.5 Summary of regional risk assessment

The results from the regional risk assessment show that pharmaceuticals are the stressors that cause the highest risk within the study area. However, the data that have been used to derive the result contain uncertainties. The uncertainty of the literature data can be concluded as being high since the data is not site-specific. By estimate, the uncertainty is approximately a factor 10. For the model calculations, there are uncertainties but the results are likely to be more correct than the concentrations from the literature. The relative risk assessment shows that the subareas around Trollhättan and Älvängen are of highest relative risk. These results can be considered as having a relatively low uncertainty based on the results from the sensitivity analysis, which showed that there are no major changes in the result when altering the input parameters.

6 Discussion

The overall results of this risk assessment show that that there is a risk of adverse effects on the ecosystem due to the emissions of stressors within the river basin. This situation is confirmed by both of the two risk characterisation methods used. Though, there are large uncertainties in the results and consequently there is a need to confirm the results by site-specific measurements and/or more detailed modelling. In the following sections, these results will be discussed in more detail.

The results from the PEC/PNEC risk characterisation show that the predicted environmental concentrations for some of the stressors are higher than the predicted no effect concentration. Consequently there is a risk for adverse ecological effects to occur. The group of stressors that indicate the highest risk are the pharmaceuticals, both measured data and model calculations indicate that the concentrations of these substances in Göta älv are at a level that can cause negative effects on the aquatic ecosystem. The calculated concentrations of pharmaceuticals are probably relatively close to the actual environmental concentrations. That statement is based on the assumption that the prescription of pharmaceuticals, the age distribution of the public and the percentage of wastewater from hospitals to the WWTPs are relatively homogenous in Sweden. Thereby, it is possible to use effluent concentrations from other Swedish WWTPs to calculate good environmental concentrations when considering connected PE, treated volume of wastewater and dilution. Among the pharmaceuticals, propranolol and carbamazepine were the substances that exhibited the highest risk while the risk posed by diclofenac and ethinylestradiol appear to be slightly lower. All of these four substances have the potential to cause adverse effects to the aquatic ecosystem. The results from this risk assessment indicate that there may be a need to reduce the risk to the environment posed by the pharmaceuticals. Still, site-specific measurements within the river basin of Göta älv are recommended to confirm the results.

The results from the PEC/PNEC risk characterisation for the organic pollutants and the pesticides were not conclusive. The measured data indicated that the predicted environmental concentrations were higher than the predicted no effect concentrations while the model calculations revealed the opposite. The discrepancies between the results can be due to spatial differences, such as different dilution of the stressors. Göta älv is the largest river in Sweden with regard to water flow, and the stressors can therefore be expected to be more diluted in Göta älv than in other Swedish watercourses. Another reason for the discrepancies can be that the model calculations did not consider all relevant stressor sources due to lack of data. This is especially true for the organic pollutants. The measured data can therefore be expected to overestimate the environmental concentrations while the model calculations underestimate them. The actual predicted environmental concentrations for these groups of stressors probably lie between the model calculations and the measured data. In order to confirm this theory, site-specific measurements are needed.

For the metals, in contrast to many of the other studied stressors, there is a large amount of site-specific data available. Today, the concentrations of the studied metals are close the background concentrations (SLU, 2009a; SLU, 2009b; Naturvårdsverket, 2000) and the PEC/PNEC ratios are below one. The risk of metals causing adverse effect to the aquatic ecosystem can be considered as low. Since metal concentrations have been measured for a long period of time in Göta älv, the dataset used to derive these conclusions can be considered as both adequate and reliable.

The predicted environmental concentrations discussed above have been calculated for the scenario that the stressors have been completely mixed with the water in the river, and have thereby been diluted to its maximum. Before complete mixture has been achieved, the concentration of stressors will be higher than the general concentrations calculated at maximum dilution.

Based on the densities of stressor sources within the different subareas of the river basin, high-risk areas were identified using the relative risk model. It should be noted that the relative risk assessment does not consider the increasing concentrations in the river as it flows towards Göteborg. The RRM only considers the connection between sources and habitats within a subarea. Due to that, the increasing risk downstream that was noted in the model calculations cannot be seen in the RRM results. The results from the relative risk assessment showed that there are two areas that are of higher risk than the others, a subarea located at Trollhättan and one located at Älvängen. In these two areas, there is a relatively high density of many of the stressor sources. These two areas each contain a relatively large WWTP, which substantially contributed to the high risk ranking of these areas. All areas containing a WWTP have been given moderate or high relative risk ranking, indicating that WWTPs have a large impact in the risk ranking. The areas that were ranked as being of no risk obtained that rank since there are no habitats of salmon or trout located within those subareas. If the endpoint were to be changed, the results of this relative risk assessment may change substantially. Since WWTPs have been shown to have a large impact on the risk, risk reduction measured should be aimed at reducing the amount of stressors emitted from WWTPs. That can be achieved either by improving the WWTPs capacity to remove stressors from the wastewater or by reducing the amount of stressors that reach the WWTPs.

As I see it, the most sustainable solution to this problem is to reduce the amount of stressors that reach the WWTP by decreasing the use of pharmaceuticals and chemicals in the society. That is because the increasing use of chemicals in the society inherently causes an increased risk to the environment. Substances that are persistent and designed in a way that makes them hard to degrade in the environment should not be emitted to the environment. For substances, which it is difficult to reduce the use of, another solution may be to reduce the toxicity so that they have less negative impact when released into the environment. By reducing the use or toxicity of these substances, the problem is solved at the source of the problem in comparison to reducing the problem downstream, as would be the case by improving the capacity of WWTPs to remove pollutants. How to accomplish this is a difficult question for which an investigation of its own is necessary. Some possible solutions though can be to place larger demands on the chemical producers to know more about the substances that they produce but also to reduce the toxicity of the substances if possible. By applying the precautionary principle to the use of chemicals, the ecological risks can be reduced and the health of the ecosystems can be safeguarded. Another solution may be to have a more strict chemical regulation that prevents persistent chemicals from being emitted into the environment. For some of the stressors studied here, for example the organic pollutants and the metals, REACH can work as such a regulation. But, REACH does not include pharmaceuticals and therefore it may be necessary to create an additional chemicals regulation that also includes pharmaceuticals since they are of environmental concern, as this study indicates.

What type of adverse effects that can be expected to occur in the aquatic ecosystem in Göta älv are difficult to predict based on the toxicity studies that have been performed so far for the stressors in this study. Most toxicity tests that have been performed for the stressors are acute toxicity tests where death has been the endpoint. Based on these tests, it is very difficult to say

what the effects would be when an organism is exposed during long periods of time. It is also difficult to predict the no effect concentration with high certainty when the toxicity dataset is not complete. As a result, the PNEC values used in this study are uncertain and effects can occur both at concentrations higher or lower than the predicted no effect concentrations. The PEC/PNEC ratios calculated in this study should therefore be considered as indicative, since both PEC and PNEC contain uncertainties. Another factor that affects the PEC/PNEC ratios is the fact that in Göta älv, there is a mixture of stressors present. Depending on which stressors that that are in a mixture, the effects can vary. When considering interactions between effects caused by a mixture of stressors, it is possible that effects occur below the predicted PNECs for the separate stressors.

The PETAR procedure has been proven to be applicable to the study area of Göta älv since it has generated results both regarding risk ranking of the different subareas within the study area but also information regarding potential concentrations of stressors in the river. Though, if more data was available, the results of the method could be more reliable. Compared to the MIFO procedure that is used for the inventory of contaminated sites today, the PETAR procedure has advantages. Since the PETAR procedure considers entire regions, high-risk areas within a region can be identified, and the subareas can be ranked with regard to remediation need. By doing so, a more efficient risk reduction process can be achieved. It would be beneficial to apply some type of regional perspective into the MIFO procedure so that an environmental and economically efficient risk management process can be obtained.

Based on this study, some recommendations can be made so that the situation within the Göta älv river basin can be better characterised with regard to ecological risks. The current information about the situation in Göta älv is insufficient for making adequate and reliable risk assessments for the river basin. In order to reduce the uncertainties, measurements of environmental concentrations of the stressors need to be performed so that site-specific information is available. By constructing a more detailed emissions model for the area, the measured data can be confirmed and the stressor sources that emit the largest amounts of stressors can be identified. This information can then be used to guide the risk management process for the river basin so that it is performed in an efficient manner with regard to both economics and achieved effects. Further, it is also preferable to have more toxicity data for chronic exposure available, since that would reduce the uncertainty linked to the predicted no effects concentrations. A final recommendation relates to the chemicals use within the society today and the responsibility of the producers of persistent chemicals. For substances such as pharmaceuticals, environmental fate must also be taken into consideration when designing the substance. The focus cannot only be on benefits for the user, such as not having to take as many tablets if the tablet takes longer time to degrade. For all substances that will be emitted to the environment at some stage of the life cycle, potential environmental effects must be researched before permission to sell the substance on the market is granted.

7 Conclusions

This study indicates that there is a general risk of adverse ecological effects for salmon and trout in the river basin of Göta älv caused by pharmaceuticals. Though, there are uncertainties in the results mainly due to the lack of site-specific measurements for many of the stressors.

Pharmaceuticals were identified as being the group of stressors that poses the highest risk to the aquatic ecosystem. For the pharmaceuticals, the predicted environmental concentrations were up to seven times higher than the predicted no effect concentrations. There are large uncertainties within these results and therefore it is recommended that these substances should be studied further within the study area so that the uncertainties can be reduced. The metals on the other hand showed no risk since the measured concentrations are well below the PNECs. The results of the metals are more certain than the results of the other studied stressors since there is a large amount of site-specific data available.

Further, the study also identified the two subareas at Trollhättan and Älvängen as being high-risk areas to the salmon and trout populations in relation to the risk of the other subareas within the river basin. The assessment shows that the reason for this is the WWTPs that are located in these subareas, which have large impacts on the risk to the aquatic ecosystem.

The uncertainties in the results from the relative risk model are considered lower than the results from the PEC/PNEC risk characterisation since the datasets used for this risk characterisation were more complete. The sensitivity analysis showed slight changes in the risk ranking when changing the input parameters.

Finally, the PETAR procedure has been assessed as being applicable to a study area as the river basin of Göta älv. The results from this study are uncertain due to limited amounts of site-specific data. As a next step, it would be recommended to also perform the last tier of the PETAR procedure, which is the local risk assessment. By doing so, site-specific data can be collected and the uncertainties of the results can be reduced. Further, risk reduction methods may be necessary for the pharmaceuticals since the results indicate that the concentrations are possibly high enough to cause adverse ecological effects. A more detailed study would also clarify the need of this.

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Appendix

Appendix A – EMIR facilities

Anläggnings-nummer	Branschkod	Anläggning	Tillsynsmyndighet
1440-1007	90.10	Älvängens avloppsreningsverk	Länsstyrelsen Västra Götaland
1440-11000	90.60	Redox Bilfarm	Ale kommun
1440-1101	24.10	Eka Chemicals AB	Länsstyrelsen Västra Götaland
1440-1113	24.10	Perstorp Oxo AB, Ale	Länsstyrelsen Västra Götaland
1440-1131	24.10	Axel Christiernsson AB	Ale kommun
1440-1145	28.20	Cogra Pro AB	Ale kommun
1440-1146	74.10	Kungälvs Fatrenovering AB	Ale kommun
1440-1147	34.80	LB´s Mekaniska Verkstad AB	Ale kommun
1440-1148	63.10	Surte Åkeri AB	Ale kommun
1440-1149	39.60	Cemex AB	Ale kommun
1440-1150	99.99	Ales avloppsledningsnät till RYA	Ale kommun
1440-1151	90.310	Skanska, Ale	Ale kommun
1440-1152	24.110	Aero-Spray AB	Ale kommun
1440-1155	90.130	Banverket Jakobsdal	Ale kommun
1440-1301	90.300	Sörmossens avfallsupplag	Länsstyrelsen Västra Götaland
1440-1401	40.60	Nödinge panncentral	Ale kommun
1462-1001	90.10	Lilla Edets avloppsreningsverk	Länsstyrelsen Västra Götaland
1462-1103	21.10	Edet bruk	Länsstyrelsen Västra Götaland
1462-1107	21.10	Inlands Kartongbruk	Länsstyrelsen Västra Götaland
1462-1121	24.110	Askania AB	Lilla Edets kommun
1462-1123	1.20	Ola Eriksson	Lilla Edets kommun
1462-1126	28.80	Premier Manufacturing Support Service L.P.	Lilla Edets kommun
1462-1127	63.10	Lilla Edets Industri och Fastighets AB	Lilla Edets kommun
1462-1128	63.10	Lödöse Varf och Näringspark	Lilla Edets kommun
1462-1301	90.160	Högstorps avfallsupplag	Länsstyrelsen Västra Götaland
1480-1104	90.290	Tagene avfallsupplag	Länsstyrelsen Västra Götaland
1480-1120	74.10	Göteborgs Emballageservice AB	Göteborgs Stad
1480-1141	28.10	Ferroprodukter AB	Göteborgs Stad
1480-1146	28.10	Provexa AB	Göteborgs Stad
1480-1147	39.50	V-TAB AB	Göteborgs Stad
1480-1152	24.140	Apoteket Produktion & Laboratorier AB	Göteborgs Stad
1480-1170	24.110	Geveco Industri AB	Göteborgs Stad
1480-1171	39.20	International Färg AB	Göteborgs Stad
1480-1174	39.20	Tefco AB	Göteborgs Stad
1480-1195	40.50	Angereds Värmecentral	Göteborgs Stad
1480-1210	15.300	Estrella, Angered	Göteborgs Stad
1480-1212	90.70	Renova, Skräppekärr sorteringsanläggning	Göteborgs Stad
1480-1228	24.110	Caparol Sverige AB	Göteborgs Stad
1480-1229	90.100	Skrotfrag AB	Göteborgs Stad
1480-1240	25.20	Powerpipe	Göteborgs Stad
1480-1241	25.40	Texla Industri AB	Göteborgs Stad
1480-1243	24.110	Elastogran Nordic AB	Göteborgs Stad
1480-1273	90.160	Renova Marieholm	Göteborgs Stad
1480-1303	39.20	Rodlin-Sanco AB	Göteborgs Stad
1480-1319	93.20	Kvibergs krematorium	Göteborgs Stad
1480-1329	63.10	Tibnor AB, hamnverksamhet kaj 262	Göteborgs Stad
1480-1333	28.70	Duroc Engineering i Göteborg AB	Göteborgs Stad
1480-1350	40.51	Backa Panncentral	Göteborgs Stad
1480-1362	63.10	Akzo Nobel Salt AB	Göteborgs Stad

1480-1364	90.70	Sita Sverige AB	Göteborgs Stad
1480-1415	24.140	Carls-Bergh Pharma AB	Göteborgs Stad
1480-1417	24.110	Peroxide Propulsion	Göteborgs Stad
1480-1418	90.40	Alelyckan Återvinningscentral	Göteborgs Stad
1480-1422	90.40	Renova, Tagene ÅVC	Göteborgs Stad
1480-1425	63.10	Kungsleden AB	Göteborgs Stad
1482-1127	24.110	Lanat Plast	Kungälvs kommun
1482-1130	90.310	NCC Roads AB Skälebräcke	Kungälvs kommun
1487-1120	27.70	Vargön Alloys AB	Länsstyrelsen Västra Götaland
1487-1165	63.30	Trollhättan Vänersborgs Flygplats	Länsstyrelsen Västra Götaland
1487-1401	40.50	Värmecentral Önafors	Vänersborgs kommun
1488-1001	90.10	Trollhättan avloppsreningsverk,Arvidstorp	Länsstyrelsen Västra Götaland
1488-1101	27.130	Stena Recycling AB, Trollhättan	Trollhättans Stad
1488-1104	34.60	Volvo Aero Corporation	Länsstyrelsen Västra Götaland
1488-1112	34.60	Saab Automobile AB	Länsstyrelsen Västra Götaland
1488-1130	90.440	Industrial Quality Recycling	Länsstyrelsen Västra Götaland
1488-1137	28.50	Trollhättans Metallsprutn. AB	Trollhättans Stad
1488-1145	24.10	Eka Chemicals AB	Länsstyrelsen Västra Götaland
1488-1146	34.70	Parker Hannifin AB	Trollhättans Stad
1488-1148	90.440	Svensk Kabel- och Metallgranulering AB	Länsstyrelsen Västra Götaland
1488-1149	34.70	Finnveden Powertrain AB	Trollhättans Stad
1488-1152	90.70	Hans Andersson Recycling Göteborg AB	Trollhättans Stad
1488-1301	90.340	Munkebo fd industriavfallsupplag	Länsstyrelsen Västra Götaland
1488-1401	40.40	Stallbacka Kraftstation Block 1 & 2	Länsstyrelsen Västra Götaland
1488-1402	40.51	Värmecentral Kronogården	Trollhättans Stad
1488-1403	40.50	Stallbacka Värmeverk	Trollhättans Stad
1488-1404	40.50	Värmecentral Lextorp	Trollhättans Stad
1488-1405	93.20	Trollhättans krematorium, Håjums begravningsplats	Trollhättans Stad

Source: Länsstyrelsen Västra Götaland, 2005b

Appendix B – Wastewater treatment plants

Anläggnings-nummer	Branschkod	Anläggning	Tillsynsmyndighet
1487-1001	90.10	Vänersborgs avloppsreningsverk, Holmängen	Vänersborgs kommun
1488-1001	90.10	Trollhättan avloppsreningsverk, Arvidstorp	Länsstyrelsen Västra Götaland
		Hjärtum avloppsreningsverk	Lilla Edets kommun
1462-1001	90.10	Lilla Edet avloppsreningsverk	Länsstyrelsen Västra Götaland
		Nygårds avloppsreningsverk	Lilla Edets kommun
		Lödöse avloppsreningsverk	Lilla Edets kommun
1440-1007	90.10	Älvängens avloppsreningsverk	Länsstyrelsen Västra Götaland
		Diseröds avloppsreningsverk	Kungälvs kommun

Appendix C – Landfills

Anläggnings-nummer	Branschkod	Anläggning	Tillsynsmyndighet
1488-1301	90.340	Munkebo fd industriavfallsupplag	Länsstyrelsen Västra Götaland
1462-1301	90.160	Högstorps fd avfallsupplag	Länsstyrelsen Västra Götaland
		Gårdstens fd avfallsupplag	
		Rösereds fd avfallsupplag	
		Bönereds fd avfallsupplag	
		Ekereds fd avfallsupplag	

Appendix D - MIFO objects

ObjektID	Objekt	Kommun	Bransch
F1440-0001	Höganäs-Bohusverken	Ale	Sekundära metallverk
F1440-0002	Preem Grönnäs	Ale	Bilvårdsanläggning, Bensinstation
F1440-0005	OK Väst AB, Nödinge	Ale	Bilvårdsanläggning, Bensinstation
F1440-0006	Preem Bohus	Ale	Bilvårdsanläggning, Bensinstation
F1440-0007	Statoil, Surte	Ale	Bilvårdsanläggning, Bensinstation
F1440-0009	Bohus Varv fd	Ale	Varv
F1440-0011	Wicanders	Ale	Verkstadsindustri, Övrigt
F1440-0014	Ahlafors Spinnerifabrik	Ale	Textilindustri, Färgindustri, Verkstadsindustri, Övrigt
F1440-0015	Shell Älvängen	Ale	Bilvårdsanläggning, Bensinstation
F1440-0016	Tudors gamla avfallsupplag	Ale	Övrigt
F1440-0017	Färgfabrik Älvängen	Ale	Färgindustri
F1440-0024	Fast Grävmaskiner HB	Ale	Verkstadsindustri, Bilvårdsanläggning
F1440-0025	Ledu fd. Wasoverken (Fd finmekanisk verkstad)	Ale	Ytbehandling av metaller, Verkstadsindustri, Övrigt
F1440-0034	Jennylunds ridklubb	Ale	Övrigt
F1440-0101	EKA Chemicals AB	Ale	Kloralkali
F1440-0102	Surte östra industriområde	Ale	Glasindustri, Bilvårdsanläggning, Verkstadsindustri
F1440-0106	Folke Stigens Fabriks AB	Ale	Ytbehandling av metaller
F1440-0110	Elmek AB	Ale	Verkstadsindustri, Ytbehandling av metaller
F1440-0112	Tudor AB	Ale	Ackumulatorindustri
F1440-0113	Neste Oxo AB	Ale	Övrig organisk kemisk industri
F1440-0114	Neste-Polyester	Ale	Övrig organisk kemisk industri, Tillverkning av plast-polyester, Verkstadsindustri
F1440-0131	AB Axel Christiernsson	Ale	Övrig organisk kemisk industri, Sjötrafik-Hamnar
F1440-2001	Rapenskår	Ale	Kommunal avfallsdeponi
F1440-2003	Valås ("Starrkärrstippen")	Ale	Kommunal avfallsdeponi
F1440-2013	Tidermans	Ale	Sjötrafik-Hamnar, Övrigt
F1440-2016	Alafors 1:2	Ale	Industrideponi
F1440-3001	Surte hamn	Ale	Sjötrafik-Hamnar, Bilvårdsanläggning, Glasindustri
F1440-3003	Nödinge båtklubb	Ale	Sjötrafik-Hamnar
F1440-3004	Nol båtklubb	Ale	Sjötrafik-Hamnar
F1440-3005	Tollered småbåtshamn	Ale	Sjötrafik-Hamnar
F1440-3006	Älvängens företagsby	Ale	Ytbehandling av metaller, Sjötrafik-Hamnar, Övrigt
F1462-0001	Hercules	Lilla Edet	Övrig organisk kemisk industri
F1462-0010	AB Mirallbåtar	Lilla Edet	Varv
F1462-0014	Torskogs Varv AB/ Thorskogs mekaniska AB	Lilla Edet	Varv, Verkstadsindustri, Gjuteri
F1462-0020	SPIMFAB: Gulf	Lilla Edet	Bensinstation
F1462-0101	Göta Cellulosa AB, också kallat Göta bruk	Lilla Edet	Massa och pappersindustri
F1462-0103	Edet Bruk, SCA Hygiene Paper AB	Lilla Edet	Massa och pappersindustri
F1462-0107	Inlands Kartongbruk	Lilla Edet	Massa och pappersindustri
F1462-0108	Lödöse Varv	Lilla Edet	Varv, Sjötrafik-Hamnar, Verkstadsindustri, Ytbehandling med lack, färg eller lim, Övrigt
F1462-0120	VYAB	Lilla Edet	Ytbehandling av metaller
F1462-3001	Göta Hamn, Lilla Edet	Lilla Edet	Sjötrafik-Hamnar
F1462-3002	Vinteruppställning Lilla Edet	Lilla Edet	Sjötrafik-Hamnar

F1480-0031	Skrotfrag	Göteborg	Bilfragmentering
F1480-0041	Dorch Bäcksin	Göteborg	Färgindustri
F1480-0044	Bachero Taicumer f.d.	Göteborg	Bilskrot och skrothandel, Träimpregnering
F1480-0052	Svetsmekano-Pulmax/ Gränges Metalock AB	Göteborg	Ytbehandling av metaller, Verkstadsindustri
F1480-0053	Bjercke Standard Varnish	Göteborg	Verkstadsindustri, Färgindustri
F1480-0054	Geveko Industri AB	Göteborg	Färgindustri
F1480-0055	Wockatz	Göteborg	Bilskrot och skrothandel
F1480-0063	Färg AB International	Göteborg	Färgindustri
F1480-0064	Västsvenska Lantmäns förening	Göteborg	Övrigt
F1480-0066	Göteborgs-Dals Pappersbruk	Göteborg	Massa och pappersindustri
F1480-0067	Gamlestadens Pappersbruk	Göteborg	Massa och pappersindustri
F1480-0071	Färg AB International f.d , Orrekulla	Göteborg	Färgindustri, Övrig oorganisk kemisk industri
F1480-0072	JH Bildemontering, Troedssons fanerfabrik f.d. Mfl	Göteborg	Bilskrot och skrothandel, Bilvårdsanläggning, Övrigt
F1480-0073	Göteborgs Siporexfabrik f.d./SAKAB/div småindustri	Göteborg	Bilvårdsanläggning, Bilskrot och skrothandel, Anläggning för miljöfarligt avfall, Övrigt
F1480-0074	Skeppsupphuggning Göteborg AB	Göteborg	Bilskrot och skrothandel
F1480-0076	Göteborgs Bilskrot AB	Göteborg	Bilskrot och skrothandel
F1480-0077	Sverres Varv (JT Isolering AB) f.d	Göteborg	Varv
F1480-0079	Kärra Bil o Plåt AB f.d.	Göteborg	Bilvårdsanläggning
F1480-0080	Balatum AB f.d.	Göteborg	Textilindustri, Bilvårdsanläggning, Fotografisk industri
F1480-0081	Trämjölsfabriken f.d., Div. små verksamheter	Göteborg	Bilvårdsanläggning, Övrigt
F1480-0084	Göteborgs Färg och fernissfabrik	Göteborg	Färgindustri
F1480-0085	Göteborgs Träimpregneringsverk Förening f.d.	Göteborg	Träimpregnering
F1480-0086	Stena Gotthard Återvinning AB	Göteborg	Bilskrot och skrothandel
F1480-0088	Bröderna Edstrand	Göteborg	Verkstadsindustri
F1480-0092	M. L. Wittboldts f.d. /Beijer Byggmaterial AB	Göteborg	Träimpregnering
F1480-0093	Ferroprodukter AB	Göteborg	Ytbehandling av metaller
F1480-0094	Tibnor f.d.	Göteborg	Ytbehandling med lack, färg eller lim
F1480-0097	G.E.Petterssons Prårnvarv f.d.	Göteborg	Varv
F1480-0098	Oklands ås, Bönereds industriomr.	Göteborg	Industrideponi, Bilskrot och skrothandel, Bilvårdsanläggning, Övrigt
F1480-0100	Svensk Bilsanering, Tagene Beg. Bildelar f.d.	Göteborg	Bilskrot och skrothandel
F1480-0136	Original Odhner AB	Göteborg	Bilvårdsanläggning, Färgindustri, Verkstadsindustri, Ytbehandling av metaller
F1480-0137	Marieholms bilskrot	Göteborg	Ytbehandling med lack, färg eller lim, Bilskrot och skrothandel, Bilfragmentering
F1480-0138	Oljehamn i Marieholm	Göteborg	Oljedepå, Sjötrafik-Hamnar, Varv, Övrigt

	(del av), Sv Eng Min.olje AB		
F1480-0139	Kemikalie- och oljedepå Marieholm (del av)	Göteborg	Oljedepå, Bilvårdsanläggning, Övrigt
F1480-0146	Tvättman Spadegatan	Göteborg	Kemtvätt
F1480-0174	Ascon Kemi Ab f.d.	Göteborg	Tillverkning av plast- polyuretan
F1480-0176	Cedervall & Söner AB (Angered)	Göteborg	Gjuteri
F1480-0227	Hammars Kem- o. Tvättbar AB	Göteborg	Kemtvätt
F1480-0280	Tvättjänst AB Göteborgs, Kortedala	Göteborg	Kemtvätt
F1480-0300	fd Grundfos AB Angeredsvinkeln 9	Göteborg	Verkstadsindustri
F1480-0301	Söderqvist Services AB fd	Göteborg	Verkstadsindustri
F1480-0326	Gamlestadens Pappersbruk sediment	Göteborg	Massa och pappersindustri
F1480-0327	Göteborgs-Dals Pappersbruk sediment	Göteborg	Massa och pappersindustri
F1480-0341	Tvätt-mäster, kortedala	Göteborg	Kemtvätt
F1480-0389	Brukens Härdverkstäder	Göteborg	Ytbehandling av metaller
F1480-0414	Nya Wikings mekaniska verkstad	Göteborg	Ytbehandling av metaller
F1480-0419	Förenade Färg	Göteborg	Bilvårdsanläggning, Bensinstation, Ytbehandling av metaller
F1480-0608	Uddens varv (I)	Göteborg	Varv, Bilskrot och skrothandel, Bilvårdsanläggning
F1480-2007	Ekered	Göteborg	Industrideponi, Kommunal avfallsdeponi, Bilskrot och skrothandel
F1480-2008	Gårdsten	Göteborg	Bilskrot och skrothandel, Industrideponi
F1480-2013	Brandkärr	Göteborg	Industrideponi, Bilskrot och skrothandel
F1480-2014	Bönered	Göteborg	Industrideponi, Övrigt
F1480-2021	Angered 7:196	Göteborg	Industrideponi, Bilskrot och skrothandel
F1480-2027	Rösered	Göteborg	Bilskrot och skrothandel
F1482-0003	Dösebacka	Kungälv	Betongindustri
F1482-0014	Kungälvs Fatrenovering	Kungälv	Övrigt
F1482-0015	Samhall fd	Kungälv	Verkstadsindustri, Ytbehandling av metaller, Övrigt
F1482-0036	Ringön Chrome AB fd m fl	Kungälv	Ytbehandling av metaller, Verkstadsindustri, Övrig organisk kemisk industri
F1487-0015	Preem, Vargön	Vänersborg	Bilvårdsanläggning, Bensinstation
F1487-0016	OK, Vargön	Vänersborg	Bensinstation
F1487-0120	Vargön Alloys AB	Vänersborg	Ferrolegering
F1487-0121	Holmen Paper AB, Wargön Bruk	Vänersborg	Massa och pappersindustri
F1487-0152	LignoTech Sweden AB	Vänersborg	Massa och pappersindustri
F1487-4003	Rödfyr: Nygård	Vänersborg	Gruva och upplag
F1488-0001	Centralförrådet, Bensin	Trollhättan	Bensinstation
F1488-0001	Centralförrådet, Bensin	Trollhättan	Anläggning för miljöfarligt avfall
F1488-0003	OK Tunhemsvägen 16	Trollhättan	Bensinstation
F1488-0004	OK Lantmannavägen	Trollhättan	Bilvårdsanläggning, Bensinstation
F1488-0006	Jet Lextorpsvägen	Trollhättan	Ytbehandling av metaller, Bensinstation
F1488-0008	Shell Lextorpsvägen	Trollhättan	Bensinstation
F1488-0010	IF SAABs jakt o sportskytte	Trollhättan	Skjutbana
F1488-0013	Trollhättans Skarpskytteför.	Trollhättan	Skjutbana
F1488-0016	Trollhättans FBU-förening	Trollhättan	Skjutbana

F1488-0017	Sandhemsskroten	Trollhättan	Bilskrot och skrothandel
F1488-0018	Din-X, Dalhem	Trollhättan	Bensinstation
F1488-0019	Vattenfall, Olidan	Trollhättan	Elektroteknisk industri, Gjuteri
F1488-0020	Sjuntorps Textilfabrik	Trollhättan	Textilindustri, Verkstadsindustri
F1488-0024	Östergårds Bensin o Service	Trollhättan	Bilvårdsanläggning, Sågverk, Bensinstation
F1488-0025	Uno-X, Albertsvägen	Trollhättan	Bilvårdsanläggning, Bensinstation
F1488-0026	Norsk Hydro, Br Brandt i Älvsborg	Trollhättan	Bilvårdsanläggning, Bensinstation
F1488-0027	Preem, Sjuntorp	Trollhättan	Bensinstation
F1488-0028	Preem, Tunhemsvägen	Trollhättan	Bilvårdsanläggning, Bensinstation
F1488-0030	SPIMFAB: Q8 Drottninggatan	Trollhättan	Bensinstation
F1488-0033	Statoil, Karltorpsvägen	Trollhättan	Bilvårdsanläggning, Bensinstation
F1488-0042	Shell Brunnered Åsaka	Trollhättan	Bensinstation
F1488-0044	Stallbacka industriområde (Ferro)	Trollhättan	Ytbehandling av metaller, Anläggning för miljöfarligt avfall, Bilskrot och skrothandel, Ferrolegering
F1488-0059	Annies Kemtvätt HB	Trollhättan	Kemtvätt
F1488-0060	City Kemiska Tvätt & Färgeri AB	Trollhättan	Kemtvätt
F1488-0061	City Press & Kemiska Tvätt	Trollhättan	Kemtvätt
F1488-0063	Kemiska Tvättcentralen	Trollhättan	Kemtvätt
F1488-0064	Klädkultur Persson & Svedlund, Österlånggatan 33	Trollhättan	Kemtvätt
F1488-0066	Reno Kemiska Tvättindustri	Trollhättan	Kemtvätt
F1488-0067	Vic- Självkem	Trollhättan	Kemtvätt
F1488-0081	Tvättcentralen i Trollhättan AB	Trollhättan	Kemtvätt
F1488-0104	Volvo Aero Corporation	Trollhättan	Verkstadsindustri, Ytbehandling av metaller, Bilskrot och skrothandel, Kemtvätt
F1488-0112	SAAB, Malöga	Trollhättan	Verkstadsindustri
F1488-0138	Stridsberg & Biörck AB, Källstorp	Trollhättan	Verkstadsindustri, Gjuteri
F1488-0141	NOHAB	Trollhättan	Verkstadsindustri, Gjuteri, Ytbehandling med lack, färg eller lim
F1488-0145	EKA, Trollhättan	Trollhättan	Övrig oorganisk kemisk industri, Kloratindustri
F1488-3001	Vinteruppställning Trollhättan	Trollhättan	Sjötrafik-Hamnar

Appendix E – PEC data

Pharmaceuticals

Table E:1 Measured concentrations of pharmaceuticals (µg/l) from five different studies performed in Sweden.

Pharmaceuticals	Bendz *et al.*	Naturvårdsverket	Andersson *et al.*	Paxéus	Ferrari *et al.*
Diclofenac:					
Background			0.0005[1], 0.0005[1], 0,003		
WWTP effluent	0.12	0.027-0.7	Average 0.23	0.19, 0.16	
Surface water	0.12, 0.01, 0.00005[1]	0.003-0.006	0.002		
Propranolol:					
Background					
WWTP effluent	0,03	0.082-0.270			
Surface water	0.01, 0.01, 0.01				
Carbamazepine:					
Background					
WWTP effluent	1.18				0.87
Surface water	0.5, 0.45, 0.1				
Ethinylestradiol:					
Background			0.00025[1]		
WWTP effluent		0.00007-0.0006	0.04		
Surface water		0.00005-0.001			

Dilution of WWTP effluent: Average dilution 0.55 m^3s^{-1}/550 m^3s^{-1} ≈ 0.001;
Minimum dilution 0.55 m^3s^{-1}/125 m^3s^{-1} ≈ 0.005; Maximum dilution 0.55 m^3s^{-1}/1200 m^3s^{-1} ≈ 0.0005
[1] LOD/2 (LOD: limit of detection)

Bendz *et al*. (2005): The sampling occurred for two hours during a day in October 2002 at Källby wastewater treatment plant in Höje River. The surface water concentrations were measured at three different sites downstream of the WWTP: 283, 4021 and 7543 m from the effluent. During the sampling, the river discharge was 0,9 m^3/s and the WWTP discharge was 0,36 m^3/s and the dilution factor was therefore 2.5. LOD for diclofenac, propranolol and carbamazepine was 0.0001 µg/l in this study.

Naturvårdsverket (2008a): For the measurements in WWTP effluent, diclofenac was found above LOD in 78/84 samples, propranolol in 10/18 samples and ethinylestradiol was detected in 14/109. Stockholms Läns Landsting took the surface water samples during the year of 2007. LOD: ethinylestradiol <0.0005 µg/l.

Andersson et al, (2005): Background samples have been taken in three lakes: Lilla Öresjö, Tärnan and Stora Envättern where the influence from human activities are considered minor. For Lilla Öresjö, there are private drains from cottages that drain to the lake, which may have affected the results. 54 samples of WWTP effluent from different WWTPs in Sweden were taken during the study. The samples were taken during a one to two days sampling period. Ethinylestradiol was only found in one of these samples. For diclofenac, the study showed that the concentrations in WWTP effluent are lower for the southern part of Sweden than for the northern parts. The surface water sample of diclofenac was taken in Lake Vänern in the proximity of a WWTP. LOD: diclofenac <0.001 µg/l, ethinylestradiol <0.0005 µg/l.

Paxéus (2004): Concentrations of pharmaceuticals in WWTP effluent were measured at two locations in Sweden, at Ryaverken WWTP in Göteborg and at Källby WWTP in Lund.

Ferrari *et al*. (2003): Concentrations of pharmaceutical residues were measured in effluents form Ryaverken WWTP in Göteborg. LOD: carbamazepine <0.05 µg/l.

Pesticides

Table E:2 Measured concentrations of pesticides (µg/l) from a Swedish study

Pesticides	Adielsson and Kreuger, 2008	Calculated average concentrations
MCPA:		
Skrivarpsån	0.07, 1.20, 0.26, 0.19, 0.09, 0.04, Below LOD × 3	0.21
Vege å	0.02, 0.72, 0.32, 0.10, 0.08, 0.03, Below LOD × 4	0.13
Diflufenican:		
Skrivarpsån	0.02, 0.01, 0.01, 0.01, 0.01, 0.01, Below LOD × 3	0.008
Vege å	0.01, 0.01, Below LOD × 7	0.003

Adielsson and Kreuger (2008): The measurements of pesticides were performed in two rivers located in an agricultural area in Skåne, Sweden during the year of 2007. Samples were taken at ten different occasions during the year in the two rivers, and the sampling method used was instantaneous sampling. For Skrivsapsån, MCPA was found in quantifiable amounts in 6/10 samples, traces in 3/10 samples and one sample indicated no MCPA. For diflufenican, the same distribution of samples as for MCPA was found. In Vege å, MCPA was found in quantifiable amounts in 6/10 samples ant the remaining 4 samples contained traces. Diflufenican was only detected in quantifiable amounts in 2/10 samples, 7/10 showed trace amount and one sample indicated no traces at all. LOD: MCPA 0.003 µg/l, diflufenican 0.002 µg/l. To calculate the average, LOD/2 was applied to all samples where trace amounts were found. For the samples where no amounts were found, zero concentrations were assumed.

Metals

Table E:3 Measured concentrations of dissolved metals (µg/l) and metals bound to sediments (µg/kg dw) in Göta älv

Metals	SLU	Johansson and Skrapste	Naturvårdsverket
Cu:			
Dissolved conc. Vargön	1-3 (Median 2)		
Dissolved conc. Alelyckan	1-2.5 (Median 1.4)		
Background concentration			1.3
Zn:			
Dissolved conc. Vargön	2-14 (Median 3)		
Dissolved conc. Alelyckan	3-14 (Median 4)		
Background concentration			4.3
Cr:			
Dissolved conc. Vargön	0.10-1.75 (Median 0.40)		
Dissolved conc. Alelyckan	0.3-2.4 (Median 0.80)		
Background concentration			0.4
Hg:			
Dissolved conc. Vargön	0.0005-0.01 (Median 0.001)		
Dissolved conc. Alelyckan	0.001-0.022 (Median .0025)		
Background concentration			0.004
Sediment concentration		30-70	

SLU (2009a & 2009b): SLU, the Swedish University of Agricultural Sciences, measures water chemistry parameters in lakes and watercourses each month at several locations in Sweden. Here, data from the two stations Vargön in Vänersborg and Alelyckan close to Göteborg have been used. The interval represents the highest and lowest recorded concentrations during the period 1998-2008. All water samples are analyzed with ICP-MS.

Johansson and Skrapste (2003): Analysis of mercury concentrations in sediments was performed by SGU, Geological Survey of Sweden. The sampling occurred at two occasions and took place at two different locations, Bohus in 1990 and Dösebacka at 1995. The report do not assign for sampling or analysis methods.

Naturvårdsverket (2000): Background concentrations of metals for major watercourses in the southern part of Sweden. Increasing flow generally infer higher background concentrations.

Organic pollutants

Table E:4 Measured concentrations of organic pollutants (µg/l) in surface water, sediments (µg/kg dw) and biota (µg/kg ww)

Organic pollutants	Göta älvs vattenvårdsförbund	Johansson and Skrapste	Sternbeck *et al.*	Woldegiorgis *et al.*	Bossi *et al.*
PAH:					
Surface water	0.000013				
HBCD:					
Surface water					
WWTP effluent			0.031		
PFOS:					
Background				0.002-0.008	
Urban water				0.012-0.039	
WWTP effluent				0.0067-0.059	<0.0015-1.1
Biota				1.2-98	9.5-156

Göta älvs vattenvårdsförbund (2007): The sample is taken at Ringön in Göteborg during 2007 with a passive sampler. The sampling and analysis was performed by Göta älvs vattenvårdsförbund in cooperation with county board of Västra Götaland. The sampling site is located outside of the study area, in the Göteborg harbour. Concentrations within the study area are probably lower than this measured concentration since there are many point sources of PAHs in the harbour.Johansson

Sternbeck *et al*. (2001): The effluent sample was taken at a washing facility. The sediment samples were taken in the Stockholm area at six different locations and at different depths as well as in the river Viskan in Halland at six different locations. A total of The Stockholm samples represent urban concentrations and the samples in Viskan represent samples in close connection to a point source.

Woldegiorgis *et al.* (2006): This study is a part of the Swedish National Screening Programme of 2005. Environmental background concentrations were sampled in three reference lakes where the influence from human activities was considered minor. The urban concentrations were sampled in central Stockholm. WWTP effluent was sampled from 11 municipal WWTPs at different locations in Sweden. Biota samples came from the three reference lakes and at a marine location, Råö outside of the Swedish west coast. LOD: PFOS <0.0002 µg/l

Bossi *et al.* (2008): The samples were taken in effluent water from six municipal WWTPs and four industrial WWTPs in Denmark. The highest concentrations were measured in effluent from one of the industrial WWTPs. Biota samples were taken at four different locations, three of them were located close to the WWTPs and the fourth was located further out in the ocean. Each fish sample consisted of a pool of 10 individuals.

Nutrients

Table E:5 Measured concentrations of nutrients (µg/l) in Göta älv

Nutrients	SLU
Total nitrogen:	
Vargön	500-1050 (Median 700)
Alelyckan	400-1200 (Median 700)
Total phosphorous:	
Vargön	6-26 (Median 10)
Alelyckan	6-56 (Median 20)

SLU (2009a & 2009b): As for the data on metal concentrations in water, the nutrient data have samples and analyzed by SLU. Data from the two stations Vargön and Alelyckan during the period 1998-2008 have been used. Samples have been taken each month during the period, giving a total of 120 samples. The interval represents the highest and lowest recorded concentrations during this period. The samples have been analyzed according to the Swedish standard.

Appendix F – PNEC data

Table F:1 Assessment factors to derive a $PNEC_{aquatic}$

Available data	Assessment factor
At least one short-term $L(E)C_{50}$from each of the three trophic levels of the base-set (fish, Daphnia and algae)	1000
One long-term NOEC (either fish or Daphnia)	100
Two-long-term NOECs from species representing two trophic levels (fish and/or Daphnia and/or algae)	50
Long-term NOECs from at least three species (normally fish, Daphnia and algae) representing three trophic levels	10
Species sensitivity distribution (SSD) method	1-5
Field data or model ecosystems	Case-by-case

Source: European Chemicals Bureau, 2003

Table F:2 Assessmet factors to derive a $PNEC_{sediment}$

Available data	Assessment factor
One long-term test (NOEC or EC_{10})	100
Two long-term tests (NOEC or EC_{10}) with species representing different living and feeding conditions	50
Three long-term tests (NOEC or EC_{10}) with species representing different living and feeding conditions	10

Source: European Chemicals Bureau, 2003

Table F:3 PNEC values and the data used to derive them, concentrations are given as (µg/l) in surface water, (µg/kg dw) for sediments and (µg/kg ww) for biota

Stressors	PNEC	NOEC	Assessment factor	References
Pharmaceuticals				
- Diclofenac	0.1	-	-	Naturvårdsverket, 2008a
- Propranolol	0.005	-	-	Naturvårdsverket, 2008a
- Carbamazepine	0.42	NOEC: 2.1 (SSD, HC_5)	5	Ferrari *et al*., 2003
- Ethinylestradiol	0.0001	-	-	Naturvårdsverket, 2008a
Pesticides				
- MCPA	1.1	NOEC: 11 (*Lemma gibba*, 14 d)	10	Naturvårdsverket, 2008c
- Diflufenican	0.005	EC_{50}: 0.45 (*S. subspicatus*, 72 h)	100	Naturvårdsverket, 2008c
Metals				
- Cu	4	NOEC: 7.8 (SSD, HC_5)	2	Naturvårdsverket, 2008c
- Zn	8[1]	NOEC: 15.6 (SSD, HC_5)	2	Naturvårdsverket, 2008c
- Cr	3	EC_{50}: 30 (*C. Dubia*, 48 h)	100	Naturvårdsverket, 2008c
- Hg	0.1	LC_{50}: 100	1000	Naturvårdsverket, 1999
- Hg (sediments)	130	LC_{50}: 100	1000	Naturvårdsverket, 1999
Organic pollutants				
- PAH	0.2	LC_{50}: 200	1000	Naturvårdsverket, 1999
- PAH (sediments)	300	LC_{50}: 200	1000	Naturvårdsverket, 1999
- HBCD	0.03	NOEC: 3.1 (*Daphnia magna*, 21 d)	100	Naturvårdsverket, 2008c
- HBCD (sediments)	170	NOEC: 3.1 (*Daphnia magna*, 21 d)	100	Naturvårdsverket, 2008c
- PFOS	3	NOEC: 300 (*P. promelas*, 42 d)	100	Naturvårdsverket, 2008c
- PFOS (biota)	6[2]	TDI: 100 (rat, 2 generations)	100 and 10	Naturvårdsverket, 2008c

SSD: species sensitivity distribution, HC_5: 5th percentile of the frequency distribution, safe to 95 % of all species, TDI: tolerable daily intake

[1] For lakes and watercourses with alkalinity > 24 mg/l $CaCO_3$

[2] PNEC to protect humans, calculated according to PNEC = 0.1×TDI×body weight/daily intake of fish. Body weight (EU standard person): 70 kg, daily intake of fish (EU worst case): 0.115 kg.

Appendix G – Relative risk score matrix

	Risk scores per subarea					
Drainage area	**Pharmaceuticals**	**Pesticides**	**Nutrients**	**Metals**	**Organic pollutants**	**Total**
1	0	0	0	0	0	0
2	0	0	0	0	0	0
3	0	0	0	0	0	0
4	72	12	16	40	90	230
5	0	12	8	16	30	66
6	0	24	8	0	0	32
7	0	0	0	0	0	0
8	0	24	8	0	0	32
9	48	24	16	16	40	144
10	0	24	8	12	20	64
11	48	24	16	24	60	172
12	0	24	12	32	60	128
13	24	24	12	16	30	106
14	24	24	12	28	50	138
15	0	24	8	8	20	60
16	48	24	16	24	60	172
17	0	24	8	16	20	68
18	24	18	10	20	40	112
19	0	24	8	40	60	132
20	0	12	4	12	20	48
21	0	12	4	24	40	80
22	0	12	10	36	70	128
23	0	24	12	32	60	128
24	0	0	6	48	90	144