Use of a Toxicity Index as an Indicator for Environmental Damage to the Aquatic Environment

TEAMPEST PROJECT

Work Package 3 – Deliverable 3.1

Ficre Zehaie\(^1\), Helena Andersson\(^2\), Dennis Collentine\(^3\), Jenny Kreuger\(^2\), Yves Surry\(^1\)

\(^1\) Department of Economics
Swedish University of Agricultural Sciences, Uppsala, Sweden
Ficre.Zehaie@ekon.slu.se
Yves.Surry@ekon.slu.se

\(^2\) Department of Soil and Environment
Swedish University of Agricultural Sciences, Uppsala, Sweden
Helena.Andersson@mark.slu.se
Jenny.Kreuger@mark.slu.se

\(^3\) Department of Economics
University of Gävle, Gävle, Sweden
Dennis.Collentine@hig.se

SEVENTH FRAMEWORK PROGRAMME (FP7)
THEME 2 Food, Agriculture and Fisheries, and Biotechnology
Theoretical Developments and Empirical Measurement of the External Costs of Pesticides
Grant agreement no.: 212120
# Table of contents:

List of abbreviations ........................................................................................................................................ 3  
Preface .............................................................................................................................................................. 4  
Executive summary ........................................................................................................................................... 5  
1. Introduction and background .......................................................................................................................... 7  
2. Environmental indicators ............................................................................................................................... 9  
   2.1 Definitions and concepts .......................................................................................................................... 9  
   2.2 The use of indicators for environmental problems ............................................................................. 10  
      2.2.2 Existing circumstances .................................................................................................................. 12  
   2.3 Relative indicators .................................................................................................................................. 13  
3. Pesticide indicators ....................................................................................................................................... 15  
   3.1 Existing circumstances ........................................................................................................................... 15  
   3.2 Designing pesticide indicators – a complex processing of information .......................................... 17  
      3.2.1 Use indicators and measuring spreading and concentration ....................................................... 17  
      3.2.2 Pesticide Risk Indicator ................................................................................................................ 19  
4. Pesticide Toxicity Index ................................................................................................................................. 22  
   4.1 Construction of the Pesticide Toxicity Index ......................................................................................... 22  
      4.1.1 Use of the Pesticide Toxicity Index in Sweden today ................................................................. 24  
      4.1.2 Strengths and weaknesses of the Pesticide Toxicity Index .......................................................... 25  
5. Application of the Pesticide Toxicity Index .................................................................................................. 27  
   5.1. FOOTPRINT ....................................................................................................................................... 28  
   5.2 Validation of FOOTPRINT .................................................................................................................. 29  
      5.2.1 Swedish national monitoring programme .................................................................................... 29  
   5.3 Integrating FOOTPRINT with PTI ....................................................................................................... 31  
6. Lessons learned ............................................................................................................................................ 31  
References ........................................................................................................................................................ 34  
Data sources .................................................................................................................................................... 37
Appendixes:

Appendix 1. Technical description of REXTOX, ADSCOR and SYSCOR ................. 38
Appendix 2. Technical description of HAIR ......................................................... 39
Appendix 3. Technical description of WQI .................................................................. 40

Boxes:

Box 1. TEAMPEST Description of work (DoW), WP3 Research Tasks ....................... 8
Box 2. Definitions ........................................................................................................... 11

Figures:

Figure 1. DPSIR for surface water .................................................................................. 12
Figure 2. The process of building an indicator for pesticides ........................................ 16
Figure 3. Diffuse and point sources of pesticides to water ............................................ 18
Figure 4. Pesticide Toxicity Index for surface water for four catchments in Sweden ...... 26

Tables:

Table 1. Structure of OECD Aquatic Risk Indicators ..................................................... 38
Table 2. Classification of water quality according to the Canadian Water Quality Index .... 40

Equations:

Equation 1. Pesticide Toxicity Index .............................................................................. 23
Equation 2. Potential acute risk ..................................................................................... 39
Equation 3. Potential chronic risk ................................................................................... 39
Equation 4. Water Quality Index .................................................................................... 40
### List of abbreviations

<table>
<thead>
<tr>
<th>Abbreviation</th>
<th>Description</th>
</tr>
</thead>
<tbody>
<tr>
<td>ADSCOR</td>
<td>ADditive SCORing</td>
</tr>
<tr>
<td>CCME</td>
<td>Canadian Council of Ministers of the Environment</td>
</tr>
<tr>
<td>DPSIR</td>
<td>Driving force Pressure State Impact Response model</td>
</tr>
<tr>
<td>EEA</td>
<td>European Environmental Agency</td>
</tr>
<tr>
<td>EQS</td>
<td>European Quality Standards</td>
</tr>
<tr>
<td>FOOT-CRS</td>
<td>FOOTPRINT Catchment and Regional Scales</td>
</tr>
<tr>
<td>FOOT-FS</td>
<td>FOOTPRINT Farm Scale</td>
</tr>
<tr>
<td>FOOT-NES</td>
<td>FOOTPRINT National and EU Scales</td>
</tr>
<tr>
<td>HAIR</td>
<td>HArmonised environmental Indicators for pesticide Risk</td>
</tr>
<tr>
<td>KemI</td>
<td>Swedish Chemicals Agency</td>
</tr>
<tr>
<td>OECD</td>
<td>Organisation for Economic Co-operation and Development</td>
</tr>
<tr>
<td>PSR</td>
<td>Pressure State Response model</td>
</tr>
<tr>
<td>PTI</td>
<td>Pesticide Toxicity Index</td>
</tr>
<tr>
<td>REXTOX</td>
<td>Ratio of EXposure to TOXicity</td>
</tr>
<tr>
<td>SLU</td>
<td>Swedish University of Agricultural Sciences</td>
</tr>
<tr>
<td>SYSCOR</td>
<td>SYnergistic SCORing</td>
</tr>
<tr>
<td>TGD</td>
<td>Technical Guidance Document</td>
</tr>
<tr>
<td>WP</td>
<td>Work Package</td>
</tr>
<tr>
<td>WQI</td>
<td>Water Quality Index</td>
</tr>
</tbody>
</table>
Preface

This publication is the first deliverable from work package three (WP3) in the project TEAMPEST – “Theoretical Developments and Empirical Measurement of the External Costs of Pesticides”, funded by EU seventh Framework Programme, Theme 2 – Food, Agriculture and Fisheries, and Biotechnology. The purpose of the report is to present how a toxicity index can be used as an indicator for environmental damage to the aquatic environment and to other environmental compartments.
Executive summary

The purpose of the TEAMPEST project is to identify methodologies and tools to reduce the external costs of pesticide use in agriculture. These costs include the impact on the health of agricultural workers, the environment and consumers. The aim of Work Package 3 (WP3) within the TEAMPEST project is to investigate the linkage between changes in agricultural production and environmental damage on surface waters using a pesticide indicator of environmental damage. In this report we give a background on the use of environmental indicators, including pesticide indicators, and evaluate the use of a specific indicator, the Pesticide Toxicity Index (PTI), as a preferred indicator for the effect of pesticide use on aquatic environments.

Pesticides used in agriculture may be transported from the site of application, mainly via drift, run-off and leaching, and could result in unintended effects on non-target organisms in different environmental compartments. Over the years, many of the most persistent, bioaccumulating and/or toxic compounds have been banned for use within the EU. Instead, today’s pesticides tend to be more weakly bound to soil particles, which in turn increases the risk for leaching of these compounds into streams or groundwater. Therefore, although there may be immediate environmental risks in the short run to soils and air due to high toxicity these are decreasing with time and in the long run the aquatic compartment is the environment most sensitive to the recurrent normal use of pesticides in agricultural production.

With a growing concern for environmental problems since the early 1990’s the demand for environmental indicators has increased. Indicators are measurements that at different levels of aggregation aim to simplify reality and reflect the state of environmental quality. The literature on environmental indicators in general and pesticide indicators in particular is very comprehensive and many different kinds of indicators are in use today. There are many selection criteria for an indicator. One of the most important properties of a good indicator is that it should be scientifically sound. However, this is difficult to combine with another desired property that an indicator also should be easy to use and simple to understand for the end-users. Identification and construction of a good indicator is therefore a challenging task.

The way indicators are constructed varies a great deal and depends on not only how well they represent the target environment, but also how well they serve the intended end-users. Indicators for assessing environmental risk caused by pesticides that only take into account weights and volumes of chemical compounds are not sufficient for representing associated risk for environmental damage. In recent years an increasing effort has been directed towards developing pesticide indicators that are able to assess environmental risk.

Environmental risk assessment of pesticides is a complex process; it requires good knowledge of the basic processes of pesticide fate under a variety of different environmental conditions, including the toxicity of the compounds on a range of non-target organisms. Due to the large number of pesticides on the market, and the widespread uses of many of these, the amount of information included in and represented by a pesticide indicator could be huge. It is therefore not surprising that many pesticide risk indicators have problems combining user friendliness and scientific soundness. Many pesticide indicators tend to be quite complicated and although potentially very accurate, they are limited by data availability and many end-users find them difficult to use. In this report we suggest the use of a risk indicator based on the Pesticide Toxicity Index (PTI). It is in a simple, transparent and user friendly way able to convey information on environmental status. In this index, the concentration of each pesticide is
calculated relative to its threshold value. The threshold value is defined as the highest concentration level at which the substance has likely no toxic effects in the aquatic environment. Quotients are calculated for each substance in relation to its threshold value, with quotients for all substances then summed up to one value – the Pesticide Toxicity Index. A high index value indicates a potentially higher risk for negative effects in the aquatic environment.

The PTI has been used in Sweden for the past several years both as an indicator for the state of the aquatic environment with respect to chemical compounds and for evaluating progress on a Swedish national environmental policy objective – A non-toxic environment. The index is calculated by using monitoring data from the Swedish national monitoring programme and nationally derived threshold values. While the Swedish calculations of the PTI are based on threshold values developed in Sweden, it is possible to use the same methodology in other European countries by replacing these values with EU Environmental Quality Standards (EQS). However, there is a general lack of consistent, long-term, monitoring data on pesticide concentrations in the environment. This is best compensated by the use of mathematical models developed for assessing the environmental fate pesticides.

Modelling is a substitute for intensive monitoring and use of an appropriately calibrated model makes it possible to study the impact of pesticide use on the environment in larger areas or entire countries. Models for this type of use have been developed within the FOOTPRINT program – a research project funded by the European Commission as part of the 6th Framework Programme for Research and Technological Development (FP6). FOOTPRINT is a well elaborated project with a user-friendly model and a transparent database that calculates concentrations of chemical substances in water. Using the output from the FOOTPRINT model (concentrations of different pesticide found in surface water) as input in the PTI will make it possible to describe the potential risk to the aquatic environment from different agricultural management practices. This link between the environment and the cost of management practices will then be used in other tasks in the TEAMPEST project; in WP3 to evaluate the costs of achieving environmental policy targets and in WP8 to integrate these costs into policy scenarios.
1. Introduction and background

Reducing the potential risks from the use of pesticides in agriculture is a priority in EU environmental policy. Pesticide, according to the EU directives 98/8/EC and 91/414/EEC, are biocidal and plant protection products. Plant protection products are mainly used in order to protect plants and plant products in agriculture, forestry and horticulture from weed, vermin and fungi. Biocidal products are chemical or biological pesticides that are not defined as plant protection products. The European Commission adopted a thematic strategy on the sustainable use of pesticides and a proposal on a new directive in order to decrease the potential risks of pesticide use in the EU (European Commission, 2008). The purpose of the TEAMPEST project is to identify methodologies and tools which can contribute to the development of effective policies, in the form of taxes and levies that will effectively address the external costs of pesticide use in agriculture.

Pesticide residues from pesticide use in agriculture have direct and indirect effects on society. Direct effects arise when residues directly affect humans and are a risk to human health. The indirect effects arise when damages are caused to the environment and indirectly inflict costs on human activities. This report addresses the indirect effects from the use of pesticides in agriculture as part of work package (WP3) within the TEAMPEST project. These effects will also be used in combination with the direct effects addressed by other work packages within the TEAMPEST project to evaluate policy alternatives for reducing the external costs of pesticides use.

The purpose of WP3 is to develop a methodology for estimating the costs of achieving environmental targets for pesticides. These costs in turn will then be used in WP5, WP6 and WP8 to estimate the environmental effects of alternative pesticide policies. Developing the methodology is a multiple step endeavour where each of the steps is represented by a particular task (see information in Box 1). The purpose of the first task is to study the use of indicators for environmental risk to the aquatic environment. In this report we evaluate the suitability of the Pesticide Toxicity Index (PTI) as an environmental indicator. In the literature it is commonly assumed that the use of plant protection products in agriculture is the main source of environmental risks. In this report we therefore focus on plant protection products but we follow the literature and use the term pesticide to refer to plant protection product.

Pesticide use in agriculture may have a negative impact on three types of environments; water, soil and air. In this report we evaluate the use of PTI as an aquatic environmental indicator. While the reason for this is that available databases are mainly developed for aquatic environment, the PTI can also be used for any compartment if the necessary input data are available. In addition, there are other factors which support limiting the study of pesticide effects on the environment to this compartment. In each of the three compartments there may be an effect on non-targeted organisms within that compartment or the compartment may serve as a mode for transport of contaminants to one of the other compartments. Applications of pesticides currently in use on crops may have an immediate effect on soil, water and air ecologies but the risk for long term effects on some environments has decreased substantially during recent years. Many of the very persistent, bio accumulating and/or toxic compounds are no longer registered for use within the EU. Instead, today’s pesticides tend to be more weakly bound to soil particles, which in turn increases the risk of leaching of toxic compounds into streams or groundwater. Therefore, although there may be immediate environmental risks in the short run to soils and air due to high toxicity these are decreasing.
Box 1. TEAMPEST Description of Word (DoW) WP3 Research Tasks

| Task 3.1: | Evaluation of the Use of Indicators for Pesticides in Surface Water as Proxies for Environmental Damages Associated with Pesticides Use |
| Task 3.2: | Evaluation of the Risk Effect for Determining Indicator Targets Associated with Three Types of Risk (for Events, “Normal” Conditions and Concentrations) that Arise with the Use of Pesticides in Agricultural Production |
| Task 3.3: | Estimation of the Field Level Production Costs of Alternative Beneficial Management Practices to Achieve Indicator Targets for Levels of Chemical Substances Associated with Agricultural Pesticides Found in Surface Waters |
| Task 3.4: | Scaling up Estimated Costs for Field Level Beneficial Management Practices (Task 3.3) to a Catchments (Regional) Level |
| Task 3.5: | Synthesis of Tasks into Catchments Scenarios for Policy and Program Analysis of Agricultural Pesticide Use Based on the Costs of Beneficial Management Practices (Tasks 3.3-4) and Targets for Environmental Damage Indicators for Surface Water (Tasks 3.1-2) |

with time and in the long run the aquatic compartment is the environment most sensitive to the recurrent normal use of pesticides in agricultural production.

The aquatic environment is currently a high priority issue in the EU. Both through the Water Framework Directive with requirements for water quality being implemented across EU member states and through the pesticide registration process (91/414/EEC) where water is a major concern. Out of 125 reviewed reports, 58% of the recommendations for member state strategies to mitigate the potential concern of pesticide uses were related to water (Azimonti, 2006). Reduction of environmental damages is one of the policy objectives for sustainable pesticide use. A tax or levy policy which was able to achieve environmental targets effectively would allow policymakers to directly address the environmental external costs of using pesticides. However, a method is needed for setting environmental targets and evaluating the costs of meeting those targets.

The proposed methodology requires that there be a link between crop and field management practices and the state of the aquatic environment. The PTI may serve as this link. Using the Swedish data will allow this link to be tested and then extended for use in other EU member states. The PTI is used in Sweden to facilitate monitoring and reporting of the environmental status of surface waters with respect to pesticide residuals. The index is based on an American method, developed by the U.S. Geological Survey for the American environmental monitoring programme (National Water-Quality Assessment, NAWQA). The American index has been slightly adjusted to Swedish conditions in order to be used for evaluating progress towards one of the 16 Swedish national environmental goals, A non-toxic environment (Asp and Kreuger, 2005). Development of the index has been supported by the Swedish Chemicals Agency (KemI) and is based on threshold values for about 100 plant protection agents and their decomposition compounds used by this agency (KemI, 2004 and 2008). The Swedish University of Agricultural Sciences (SLU), responsible for the Swedish Environmental Protection Agency’s ongoing pesticide monitoring program, collects and analyzes water samples from four small catchments in Sweden. Data collection for the first area started in 1990, followed by the other three areas in 2002. These data are then used to construct the PTI. These samples are complemented by extensive interviews with farmers in the four catchments.
with respect to management practices like crops, fertilization and pesticide use (type of pesticide, dosage and time). This background information is then combined with water flows, precipitation and the water samples from the areas and used to describe the state of the environment as well as to identify how it evolves over time (Adielsson and Kreuger, 2008). The documentation from these catchments will be used within TEAMPEST WP3 to evaluate the use of the PTI in environmental policy and for development of a methodology using an environmental risk indicator as a policy target.¹

The first section in the following report will review the literature on environmental indicators in general in section 2 and then focus more specifically on pesticide indicators in section 3. This is followed by an evaluation of whether the PTI can be used as an appropriate indicator for environmental damage from pesticide use. The final section of the report presents how an environmental indicator will be used in the next step; the study of three types of risk factors associated with pesticide use (Task 3.2, see Box 1). Based on the results of this step, a cost methodology or farm income optimisation model will be employed to estimate agricultural production costs associated with the adoption of alternative crop and field management practices (Task 3.3). Following this, the costs of adopting beneficial management practices on a regional level will also be estimated, based on field level cost estimates (Task 3.4). The final step is to develop policy alternatives based on the costs of achieving particular environmental targets (Task 3.5).

2. Environmental indicators

A growing concern for environmental problems has these last decades substantially increased the need of information on environmental quality. One way to satisfy this need is through environmental indicators. The objectives of deliverable 1 (D1) in WP3 in the TEAMPEST project is to investigate how this information can be provided for pesticide environmental risks. A large body of literature exists on environmental indicators in general and pesticide indicators in particular. The objectives of this section are to provide a general overview on this topic. In section 3 we present the literature on pesticide indicators.

2.1 Definitions and concepts

The literature on environmental indicators is mainly interested in definitions, concepts and methodological issues. Environmental indicators are ways of making a complex environmental reality more transparent by processing information to be easily interpreted (Bockstaller and Girardin, 2003; Niemeijer and de Groot, 2008; Riley 2001 & Donnelly et al. 2007). An indicator is made up of specifically chosen variables which give a good view of the present state of the environment and therefore also makes it possible to follow changes and trends over time (see also Box 2). These indicators may be single variables, measured or calculated within a model, of concentration of pollutants in different environments and are important for environmental policy. They are hence, for example, used in the follow-up of

¹ Note however that the definition of risk in the literature on pesticide use and environmental problems does not involve any probabilistic estimation as in other subjects. In this literature it is common to express the environmental threat as environmental risk. As Maud et al. (2001) correctly remark risk in this literature refers to “a situation that may cause harm”. In this report, in line with the literature on pesticide use and environmental problems, we will use risk as a situation that may cause harm.
national environmental objectives in Sweden. Single variables indicators from data of actual nitrogen concentration measured in the sea are for instance used as an indicator to follow up the objective of Zero eutrophication. More complex composite variables, such as indices, may also function as indicators. An index is a numerical value, made up of two or more single number variables. Single number and composite variables are in other words different ways of measuring or calculating environmental quality. They do however share the same scope of use – to indicate the state or trends of complex systems. The underlying assumption when reading an environmental indicator is therefore that it contains information on the levels, trends and underlying causes of more complex environmental systems. For instance, pesticide concentrations in a catchment are assumed to be an approximation of the environmental status of the ecosystem in the entire catchment. Designing an indicator is however associated with some difficulties as nature is complex and it is related to human activity in a complex way. In fact, it is this complexity that generates the need for indicators.

The word indicator can sometimes also be used synonymously with the word “model”, which is confusing (see also Box 2). Models can calculate variables that are difficult to observe and measure, and can be used to generate indicators. It is therefore the calculated variable in the model that is the indicator – not the model itself.

2.2 The use of indicators for environmental problems

Indicators are used to identify possible environmental problems and, as mentioned above, to fulfil national environmental objectives and therefore adjust priorities between environmental quality and other needs in a society. They may also function as inputs to other decentralized decision makers such as consumer and producers. Since the early 1990’s there has been an explosion of environmental indicators (Riley, 2001). This explosion brings with it large differences among environmental indicators and the need of structuring and harmonizing of the existing indicators. OECD was one of the first actors to call for and to systematically contribute with methodological and conceptual work to this end (OECD, 2008).

The most prominent conclusion that emerges from the literature is that indicators have intrinsic conflict that is difficult to solve. An environmental indicator is expected to be scientifically sound and simple (Niemeijer and de Groot, 2008; Bockstaller and Girardin, 2003 & Falconer, 2002). However, the meaning of what scientifically sound is, is not always clear but often it is related to its ability to correctly describe a complex environment and how it interacts with detrimental human activities. Niemeijer and de Groot (2008) interpret this among other things as indicators should cover key aspects, components and gradients of pesticides impact on the environment. To fulfil this, an environmental indicator needs to include more information, which inevitably makes the indicator more complicated. However, since the main purpose of an indicator is to simplify and make these complex interactions more understandable, there is a trade-off between the need to capture these complexities and the requirements of simplicity. Ultimately, it is a matter of balancing these two needs, taking into consideration the purpose of indicators and the existing circumstances. Below we discuss the search of this balance by referring to the literature.

Indicators related to factors putting pressure on the environment, can be used ex ante to alert potential environmental risks. Others indicators, more closely related to the effect or the state of the environment, will be more appropriated for ex post analysis to put in place correct environment policy and evaluate these policies. Early in the 1990’s OECD developed a causal chain framework to classify indicators (OECD, 1993). Environmental problems were described in a simple pressure-state-response (PSR) model. Human activities put pressure on
Box 2. Definitions

**An indicator** acts as a sign or indication of something. Both single data and calculated indices can be used as indicators in order to show the pressure, the state and/or trends of a system.

A **model** is a structured way of representing a system based on scientific knowledge and may be used to calculate environmental indicators not possible to assess or measure.

**An index** is a numerical value, made up of two or more variables. The calculated quotient can be used as an indicator in order to illustrate the environmental condition.

**A pesticide**, according to the EU directives 98/8/EC and 91/414/EEC, may be biocidal or plant protection products. Plant protection products are mainly used in order to protect plants and plant products in agriculture, forestry and horticulture from weed, vermin and fungi. Biocidal products are chemical or biological pesticides that are not defined as plant protection products. The environment and may change the state of the environment. Efforts to revert or reduce these changes are the society’s response to these changes. This model, or more elaborated versions of this framework, is frequently used in the literature to structure indicators in a casual effect chain. Alfsen and Sæbø, (1993) and Bockstaller et al. (2008) discuss the purpose of indicators related to these casual-effect frameworks. In terms of the PSR model indicators may give information on the environmental pressure, environmental state or the responses. In more elaborated versions it is possible to identify other types of indicators. The European Environment Agency (EEA) developed the PSR model further into the Driving force Pressure State Impact Response (DPSIR) model. This framework is more specific about the causes and the effect of environmental problems. Unlike the PSR model it takes into consideration the driving forces behind the pressure put to the environment. The driving forces may be population density, agriculture production and other needs in the society that result in emissions of pesticides or other pollutants. The DPSIR model also differentiates between environmental impact which refers to the state of the environment and the impact to the society (EEA, 1999). According, to this framework indicators may be constructed to give information on the driving force and the impact of environmental problems. Figure 1 presents an example of DPSIR for surface water.

The purpose of an indicator is also closely related to the intended users, referred as end-users in the literature. End-users may be consumers, producers such as farmers or their advisers, public policy makers and scientists. End-users have different needs and therefore demand different kind of indicators. Consumers or producers are private agents and need health- or production-related indicators to avoid undesired own costs, rather than indicators related to the state of the environment, which has public goods characteristics. Indicators related to the state of the environment are more interesting to policy makers to correct possible externalities. Many authors argue that a careful consideration of end-users is crucial when developing indicators (Yli-Viikari et al., 2007; Bockstaller et al., 2008; Mitchell et al., 1995; Crabtree and Brouwer, 1999; Girardin et al., 1999). Some authors argue that end-user should be more involved and active in the development of indicators (Bockstaller et al., 2008).
2.2.2 *Existing circumstances*

Existing circumstances determine the method used to build an indicator, and are therefore crucial for the choice of an indicator and its quality. Methods used to construct indicators are usually divided into measurements or calculations based on a combination of data derived from simulation models (Bockstaller et al., 2008; Riley 2001). Measurements are used when theoretical knowledge to understand the relationship between human activities and environmental quality are considered to not be good enough. This is particularly evident for very complex environmental problems such as biodiversity and sustainability. According to Merkle and Kaupenjonann (2000) these types of indicators tend to be not accurate enough. When the scientific knowledge on the process leading to environmental impact is available it is possible to improve the accurateness of indicators. More or less complicated calculations may be used to this end. Simple calculations can be used to add some relation between some variables such as underlying natural conditions, the characteristic of the environmental problem and the economic activities generating the environmental problems. Although these types of indicators are simple, there are concerns that they are poor predictors (Riley, 2001). These calculations do not consider complex systems of relations that may exist. Through models based on scientific knowledge it is possible to include at least some complex relation to calculate indicators. These models improve the prediction of environmental problems but tend to be difficult to use and are less transparent.

Other circumstances determining the use of environmental indicators is data availability. Complex models based on advanced scientific knowledge are usually data intensive. If input data are missing these indicators are not very useful. If the input data are available there are other problems to be considered. Data should cover the relevant area and be collected in a consistent way. This may be difficult if the relevant area is a large region or a country. For higher aggregation levels, such as the EU level, data need to be collected in different jurisdictions and administrative entities and the difficulties do thereby increase. We also
expect that data is available and consistent over time. Niemeijer and de Groot (2008) are more careful on this and point out that a criterion for a good indicator is how much data it requires relative to those available. If good data is available then the models can be complex and still useful. It is worth noting that as time passes, and new technologies are available more data at relatively high qualities are made available. This helps to overcome at least some of the difficulties mentioned above for complex and data intensive indicators. These indicators have however another disadvantage. They tend to be too cumbersome and too complicated for the intended end-users. Many end-users do not have the resources in terms of knowledge, time and capital to use and update complex indicators.

Not very surprisingly the nature of the environmental problem plays an important role when balancing between the degree of simplicity and scientific accuracy. The more complex the environmental problem is, the more difficult it is to combine simplicity with scientific accuracy. In case of pollution, many emitters distributed across different sectors of the economy tend to increase the complexity of the environmental problem. The problem can be further complicated if emission sources are geographically diffuse. In the same vein, the degree of complication is also related to the number and the distribution of recipient areas. In these cases, to get an indicator for the overall environmental risk one has to aggregate measures, calculations or models from different pollutants in different recipient areas at different scales. Since generally the environmental effect is related to these differences, aggregation is an important but difficult step of building an indicator. Aggregation requires consistent ways of weighting pollutants environmental effect. This requires a lot of information and good scientific knowledge on each pollutant effect on the environment and its weight relative other pollutants environmental effects.

The aggregation problem is also related to the geographical area of interests. Large geographical areas generally include large diversity of ecosystems, climates and other geophysical properties. As for the case of aggregation over different pollutants, we require consistent ways of spatial weighting of the environmental effects. As in the previous case, this spatial consistency also demands a lot of information and knowledge. Spatial aggregation over complex environmental problems poses great difficulties when designing indicators. Further challenges are added as we also require consistency over time to develop indicators intended to be used as follow ups of environmental policies.

2.3 Relative indicators

Complicated environmental problems at high aggregation levels are difficult to capture in simple ways. In this case an indicator cannot represent the large diversities of the environmental problem. It is however possible to somehow overcome these problems by constructing relative indicators. Relative indicators give information on changes and trends of the environmental problem relative a reference state. Usually the reference state is politically determined and may refer to historical environmental states or known environmental thresholds. Relative indicators need reference points to give information on how serious an environmental problem is relative the reference state. Reference values based on knowledge of environmental thresholds try to capture environmental risks, which other indicators not always do. Simple and popular types of relative indicators are indices. An index is usually a ratio between concentration of pollutants in nature and some reference values. Concentrations of pollutants are either monitored, i.e. measured in catchment areas, or calculated within a model.
Relative indicators contain important information not found in indicators related only to environmental pressure or concentrations of pollutants. The former give an indication on the risks of pollution while the latter is only a measure of natures’ exposure to pollution. Since risk captures the actual damage to the ecosystem it clearly is more interesting and is in line with the objectives of indicators. If the relative values are based on scientific knowledge helping quantifying the effects of pollutants in terms of ecological and biological losses, the indicator will fulfil its purpose and give information on environmental quality. It is then up to the society to determine the desired level of environmental quality given the damages indicated. This clearly shows the task of an indicator and separates the need of information on pollution and the economic problem of valuing these effects. There is some confusion on this issue in the literature. Falconer, (2002) discusses this issue and says that indicators ignore the relative social costs of different types of damages or damages to different ecosystems. However, well constructed indicators should only indicate environmental effects and not their values. It is however worth noting that references may be related to environmental goals determined politically and from an economic point of view this may be inefficient. Since the political process leading to these goals may be negotiation and compromises the goals are not necessarily optimal. If indicators are developed with reference values related to such environmental goals they may lead to inefficient allocations of resources.

Part of the scientific requirement for an indicator is that it needs to be validated. This may be through sensitivity analysis or testing processes. A common validation process is to see if indicators give consistent information on environmental risk across different polluters, time and space. The idea is that whenever the risk is high, the indicator should indicate this for each pollutant and different areas over time. This consistency is checked against measured pollution data or experimental data where some of these conditions are deliberately altered. A somewhat less complicated validation process may be used when indicators are calculated or come out from models. In such cases the resulting indicators can be compared to measure in some representative areas and the model is calibrated to obtain more realistic indicators. Another common validation method is to use experts that, based on their experience and knowledge, can adjust and judge the appropriateness of indicators.

There is a large literature on indicators selection criteria. However, given the objectives of the TEAMPEST project we find the criteria developed by the European Environmental Agency (EEA, 2005) to be the most relevant ones. The selection criteria require that indicators should be policy relevant, can be used to see progress towards environmental objectives and are part of the EU priorities in environmental policy. The criteria also require that the indicators are methodologically well founded to be consistent over space and time, understandable and should be available at a national scale. These are clearly practical and policy-oriented criteria, in line with the objectives of this report.

There are many environmental indicators in use and given the complexity of environmental problems this is not very surprising. In this section we have shown that this diversity depends on the existing circumstances and the purpose of the indicators. In the following section we turn to the more specific environmental indicators for pesticides.
3. Pesticide indicators

Pesticides are, according to the EU directives 98/8/EC and 91/414/EEC, divided into biocidal and plant protection products. Plant protection products are mainly used in order to protect plants and plant products in agriculture, forestry and horticulture from weed, vermin and fungi. Plant protection products are used in open systems i.e. easily interact with ecological systems and are spread to non-intended areas. Biocidal products are chemical or biological pesticides that are not defined as plant protection products. They are used in many different contexts in order to eliminate unwanted organisms such as wood preservatives, disinfectants and anti-fouling paint. Biocidal products are mainly used with great precision in the production process inside fabrics and plants and should, if used properly, not be spread to non-intended areas. Plant protection products are therefore the main source of environmental damage from the use of pesticides. We follow the literature and use the word “pesticides” throughout this report. It should however be pointed out that, since we in this report are interested in environmental problems caused by agricultural pesticide use, we mean plant protection products.

Parallel to the literature on environmental indicators, the one on pesticide indicators has expanded substantially over these last two decades (Falconer, 2002; Maud et al., 2001; Reus et al., 2002). Similar to the general literature on environmental indicators and despite the limited evidence of environmental problems associated with the use of pesticides we find many different types of pesticide indicators in use.

Figure 2 provides an overview of the process of building indicators. Related to the discussion on environmental indicators in chapter 2 we first identify some important existing circumstances conditioning the environmental problems caused by pesticides. We separate between farm-, physical- and chemical properties. The latter include only pesticide chemical properties. Physical properties are geographical and climatic properties describing the environmental conditions. Farm properties, including those in forestry and horticulture, are the economic and technological properties conditioning use of pesticides such as pesticide management practices. In section 3.2 we describe the designing of pesticides indicators. First in relation to pressure state response (PSR) we discuss *ex ante* indicators related to the use of pesticides and then other indicators more related to the environmental effect of pesticides use.

3.1 Existing circumstances

Pesticides are usually composite products and may include many active substances designed to neutralize pests. By large, the most important component of the commercial pesticide products is the active substance. These are chemical products designed to kill pests. Commercially available pesticides or compounds sometimes include more than one active substance. Each of these substances has different chemical properties, e.g. persistency, mobility, toxicity, bioaccumulation and volatilization potential. Potential environmental problems arising from the use of pesticides are therefore a result of a combination of multiple dimensions of toxicity, persistency, mobility, bioaccumulation and validation potential.
Most environmental problems are often complicated by uncertainty and thresholds. For pesticides, yet another factor that further increases the degree of complexity is their commercial use in the agriculture, forestry and horticulture sectors. These sectors produce many different kinds of products which contaminate many different environments and a large variety of active ingredients can therefore be found in one catchment area. In agriculture, plant rotation systems imply that many different pesticides are found in small areas. Further complications are added to the problem when one considers that the production of pesticides is a research-intense sector constantly feeding the market with new plant protection products.

Environmental problems caused by pesticides are numerous and dispersed over large areas. The final environmental effect of pesticides has much to do with how pesticides are transported from the emitters to recipients, how different environmental property may influence their transport and to which extent they are degraded during this process. Pesticides that end up on the ground are normally degraded by microbes. The speed of degradation depends on different factors, such as temperature, biological activity in the ground and properties of the substance. Residues that are not degraded leach out from the soil to surface or groundwater. Cracks and macro pores often present in clay soils causes fast transport down to drainage tubes and further out to watercourses. Risk of pesticide losses from these soils to surface waters are therefore higher than from soils dominated by silt or fine sand. There is also a big variation of pesticide properties. Some substances are strongly bound to soil particles and are therefore rather immobile while other substances, not so strongly bound, has a higher risk to be transported down to drainage pipes and groundwater. The environmental risk caused by the use of pesticides is usually a combination of intrinsic properties, weather, soil type, crop type and application technique. The driving force is usually precipitation (or irrigation) and the intensity of this.
Pesticides found in water may be from specific sources, sometimes in the literature referred as point sources, or be the result of diffuse leaching from larger areas (see Figure 3). The term point source sometimes describes losses from a defined spot in the farm or the field, e.g. spillage when filling the tank. The effect of point sources can be very serious and harmful to the environment since these losses often are very concentrated and occur in a limited area. These kinds of losses often happen as a result of bad management practices at the farm or in the field. In the literature of pesticides, especially in natural sciences, this is important to differentiate from other losses, referred as diffuse, that occur from the use of pesticides in larger areas, like a field. Wind drift to surrounding surface waters of pesticides sprayed on the field or pesticides that don’t degrade until they reach groundwater are two examples. These pesticide concentrations are much lower than the ones from point sources but the losses do on the other hand often derive from several hectares. However, from a policy point of view, all these emissions are diffuse since point sources of the kind described above cannot be specifically targeted. To some extent since they are random but mainly because they occur at farms or at field scales and when evaluated at policy relevant scales, such as regional or national levels, they are in fact to be considered as diffuse.

We have just discussed some properties of pesticides that make their environmental fate and effect complicated. A good environmental indicator for pesticides should be able to capture this complexity, which is the subject of the following section.

3.2 Designing pesticide indicators – a complex processing of information

Information on pesticide properties and how they are related to each other can be measured, estimated or based on some theoretical knowledge using simple calculations or complicated models. This gives information on the spread and concentration of each pesticide in nature and may be used to measure environmental risk from the use of pesticides – i.e. to develop pesticide indicators. A common way to categorise pesticide indicators is as Pesticide Use Indicators and Pesticide Risk Indicators.

3.2.1 Use indicators and measuring spreading and concentration

Pesticide Use Indicators focus on total amounts of pesticide used or frequency of application. An application occurs when pesticides are spread or applied to a fill to kill pests. The assumption is that number of application rather that the total quantity used better captures the environmental risk of using pesticides (Gravesen, 2000). Pesticide Use Indicators are widely used throughout the world and often rely on easy accessible statistics on pesticide use, quantities of pesticides applied and officially authorised doses. The Danish pesticide indicator Frequency Application (FA) is one such example. There is however no strong correlation between used volumes or application rate of pesticides and concentrations found in surface waters. Furthermore, the environmental effects of pesticides are highly dependent on their toxicity and other chemical properties and indicators based on volumes or application rate fail to capture these aspects. van Bol et al. (2003) correctly observe that volumes and application rate of pesticides in theory can be correlated to pesticide use and environmental risks if the properties of pesticides do not change and the environment exposed to these pesticides is stable. van Bol et al. (2003) did also point out that since the properties of pesticides are changing, Pesticide Use Indicators will not be able to capture the environmental risks from the use of pesticides. They also remark that Pesticide Use Indicators cannot be used across areas, regions and countries with different geographical and environmental characteristics. In
addition, temporal comparison may also be difficult as the environment due to global warming is expected to change the environment. For surface water this may be important as climate change is expected to affect precipitation and run-off in many catchment areas around the globe. Pesticide Use Indicators are therefore bad predictors of environmental risks from the use of pesticides as they do not distinguish toxicity levels and other relevant chemical properties of pesticides both at spatial and temporal scales. Indicators based on volumes and quantities of pesticides are therefore considered to be unsatisfactory (Stenrød et al., 2008; van Bol et al., 2003; Reus et al., 2002).

Adding some information on how pesticides are spread in the environment can considerably improve an indicator. From the emitters pesticides are transported to different recipient areas through air or soil. Some of the pesticides going through the soil profile are degraded during this process. The final impact to the aquatic environment depends partly on the residence time of pesticides in the recipient areas. Including this knowledge, the indicator will contain information on the environment’s actual exposure to pesticides. Indicators giving information on exposure may simply indicate the total amount of pesticides present in the environment. However, some indicators aggregate and weight pesticide risks according to their chemical properties and face difficulties associated with these steps.

The difficulties of aggregation and weighting are related to the large quantities of pesticides existing in a dynamic market of which many are spread over large areas. In addition, each pesticide has a number of properties affecting the environment. These factors make aggregation the most challenging step for pesticide indicators. Not surprisingly, the literature on pesticides, compared to the general literature on environmental indicators, devotes a lot of attention to this step. The most common way to estimate total exposure is to categorise pesticides in different groups mainly based on pesticide’s toxic properties. These categories are then given different scores depending on the relative toxicity often taking into
consideration other relevant chemical properties. More ambitious models may choose to explicitly model toxicity by using eco-toxicological or other biological and ecological knowledge. These types of indicators are complicated and tend to be less useful.

### 3.2.2 Pesticide Risk Indicator

Although indicators aggregating exposure levels are a considerable improvement from the indicators based on volumes and application rate, it is still not satisfactory enough to capture the potential risks pesticides put on the environment. To construct a risk indicator we need information on the sensitivity of recipient areas to pesticides e.g. sensitivity of aquatic organisms in waters contaminated by different pesticides. The Pesticide Impact Indicator (p-EMA) and the Pesticide Environmental Risk Indicator (PERI) are examples of indicators capturing the use of pesticides and the concentrations found in water but not their potential risk to the environment. There is however no conflict between measuring exposure and risks rather, as van Hyfte et al., 2008 point out, an appropriate risk indicator should include both exposure and the risk this constitutes to health and the environment. To develop an indicator from measuring exposure to measure risk is however yet another difficult step. A risk indicator must solve the weighting and aggregation problems and needs to distinguish different active ingredients and their chemical properties. Using additional biological, ecological and toxicological knowledge, these pesticides effect on non intended organisms must be assessed in many different cases. This requires a lot of information based on good scientific knowledge for a large combination of situations where toxicity, climatic, geographical and concentration of pesticides can vary. Despite these problems and their complexity aggregation and weighting is necessary when developing risk indicators. These last two decades have witnessed increasing efforts devoted to develop pesticide risk indicators (Stenrød et al., 2008).

The complexity of designing risk indicators for pesticides tends to give complex and therefore less user-friendly indicators. For this reason risk indicators often calculate total risk of pesticides as sums of the individual risk caused by each pesticide e.g. PTI. This is a simplification that may miss some interactions, synergies or antagonisms, among pesticides but may be reasonable given the degree of complications involved in estimating pesticide risks. Belden et al. (2007) show that linearly adding individual risk of pesticides may be a reasonable approximation with small likelihood of underestimating interactions among pesticides. In practice, the scientific knowledge on how exposure of pesticides results in environmental effects is often brought from laboratory studies. These studies calculate threshold values on how sensitive different key organisms are to concentration of pesticides. The most common way to include threshold values in indicators is by building an index where pesticide concentrations in nature are related to pesticide toxicity estimates. According to Reus et al. (2002) and Falconer (2002) indices are often preferred by scientists. One reason for that is that indices meet some basic methodological requirements on indicators in a relatively simple way. The use of threshold values in indices is relative explicit and transparent. Other ways to consider the effect of pesticide exposure in the environment is by explicitly modelling their effects on the environment. This tends to be cumbersome and less transparent for many end-users.

There is also another aggregation problem that needs to be acknowledged. The exposure and potential risks to the environment due to pesticide use are conditioned by local biological, geographical and climatic properties in different recipient areas. The larger the geographical scales are the more sensitive the indicators should be to these local conditions. This can be
solved in two different ways – either by designing flexible indicators accommodating particular circumstances such as different climatic, geographic and ecological conditions or by limiting the scope of indicators and designing specific indicators for particular climatic, geographic and ecological conditions. Today most efforts are devoted to develop sophisticated models able to give flexible indicators. Unfortunately these indicators tend to be cumbersome and not user-friendly. Accordingly little work is devoted to develop indicators specific to some areas despite the degree of complication of pesticide environmental risks and the large differences in climatic, geographic and ecological conditions for the geographical area of interest. van Bol, et al., (2003) propose to develop specific and global indicators. The former to assess more detailed issues related to the region and the latter for national and international policy. This is interesting since it points out the need of both specific and global indicators.

The specific conditions mentioned above affect threshold values and can be important determinants of pesticide risks. Threshold values may differ between different geographical areas because of differences in assimilation capacities, temperature, tolerance or history of exposure to pesticides. This may have important implication for indicators. For instance, threshold values are often calculated at national or international scales and may fail to capture regional or national differences in threshold values and thereby fail to capture regional or national differences in the risk of pesticides. This would make indices flexible enough to capture variation in exposure as concentration levels can be available at different scales but are not flexible enough to capture variation in risk. Institutional factors also hamper to correct these deficiencies. The laboratory studies, which are the foundation behind threshold values, are provided by private firms and are privately owned. Therefore, although the results of these studies are available, the laboratory tests are not available for improvements or complementary studies. These laboratory studies may also be used to improve indicators in other aspects. Falconer (2002) observes that most indicators assume that pesticide concentration increase environmental risks linearly (or additively) without any scientific motivations. Access to the laboratory studies may be helpful to find more accurate relations between indicators and pesticide concentrations. However, this is not possible as long as access to these studies is limited.

The existing pesticide indicators seem to give different conclusions about pesticide use and environmental risks. Reus et al. (2002) compared eight indicators used in Europe as part of the EU-financed project CAPER. Each indicator ranked 15 pesticide applications with different crops, land use and other environmental characteristics. The Spearman rank correlation test showed that there were large differences in the results. These findings are supported by other similar comparative studies (Maud et al., 2001; Greitens and Day, 2007; de Smet et al., 2005; Stenrød et al., 2008). Stenrød et al. (2008) use a different method to compare indicators. They used monitoring data from two catchments in Norway to evaluate the estimated environmental risks obtained from three different indicators. They found that one indicator was best because it better captured the environmental threats but also concluded that much of the differences among the three indicators is explained by how they are constructed.

The overall picture of large differences in the way pesticide indicators estimate environmental risk may be moderated if carefully investigated. One reason for these differences is that the existing indicators define the environmental problem with pesticides differently. Pesticides pose at least three different kinds of risks. The first is the heath risk to workers that may come in contact with pesticides at work. The second is health risk to consumers that may buy
pesticide contaminated products or get exposed to pesticides via drinking water and the third is the environmental risk. Which one of these risks an indicator should include seems to be arbitrary. This explains some differences in indicators. Sometimes the impression is that indicators should include all three risks. When indicators are used for policy purposes this may be misleading. From the economic theory of environmental problems and externalities we know that these risks are problems of a different nature. The first two are problems of information as both consumers and workers have the choice to avoid these risks if the risks are known and consumers and workers are correctly informed of the health risks they are exposed to\textsuperscript{2}. The environmental risk occurs when pesticides are transported to areas outside of the treated area and there possibly having a negative impact on ecosystems. This is a completely different problem as the damage do not harm those directly involved in the transaction but a third part i.e. individuals suffering from pesticide contaminated ecosystems. These individuals cannot directly choose to avoid these damages unless they indirectly can impose their choice to others through collective choices such as regulations. These different types of risks are often mixed when pesticide indicators are developed and may be confusing when pesticide indicators are used in policy.

Some differences are also explained by the intended end-users of pesticide indicators. Farmers, policy makers and scientists are the most common end-users mentioned in the literature. Reus et al. (2002) found that, out of eight different pesticide indicators, five were developed to advise farmers on pesticide use, two for policy makers and one for scientists. The resources available in terms of time and capital to end-users are important. Farmers have limited resources and prefer simple indicators and may accept less accurate indicators. Policy makers need easily interpreted indicators at higher aggregation levels and require higher degree of accuracy and probably may accept more complicated indicators. It is therefore not surprising that these choices will give different types of indicators and explains some of the differences among the indicators in use to day.

Pesticide indicators also differ in terms of the compartments included as well as the way pesticide transport is modelled (Reus et al., 2002 and Stenrød et al., 2008). Most indicators include pesticide contamination to water. Pesticide indicators for water may however include only surface water, groundwater or both. This explains some differences but again a clear definition of the problem of interest should help choose what to include. If human health is the main concern the indicator should include groundwater, which often, but not always, is used as drinking water. If the focus is on potential environmental risks posed by pesticides then the indicator should include surface water. In the literature often indicators include both surface and groundwater. Reus et al. (2002) found that seven out of eight investigated indicators for pesticides included both surface and groundwater contaminations. However note that three out of these seven give separated indicators for each compartment included. Other indicators include pesticides contaminating soil and sometimes also pesticides contamination to air. There may be synergies or antagonisms among the effects on these compartments. As many comparison studies have shown, e.g. Reus et al. (2002) and Edwards-Jones et al. (1998), this may give large differences among indicators. Furthermore, pesticide transport from emitters to recipient may also differ. Most indicators includes runoff, spray drift and other also include leaching and/or drainage. These further increase the differences between the existing indicators (Stenrød et al., 2008).

\textsuperscript{2} If consumers and workers do not have information it is either because the knowledge of the risks are not available or is asymmetrically distributed as usually producers may have the knowledge but do not reveal it. In the former case there is nothing to do until the knowledge is provided and in the latter case the most efficient policy is to regulate information rather than act through quantities or prices on the market.
As discussed in this section, pesticide indicators can be designed in many different ways based on different extent of aggregation. When pesticide indicators are intended to be used for environmental policy they need to be aggregated over large areas and need to include a large quantity of information. To construct this kind of indicator that assesses the potential environmental risk may be difficult. Probably the simplest and most transparent way to solve this problem is through indices where actual concentrations of pesticides, measured or calculated, are related to their threshold values.

4. Pesticide Toxicity Index

When using variables, either measured or calculated, as an indicator, one way to present the result is in terms of indices. An index is a numerical value, created by a ratio made up of two or more indicators or data values (Röndell, 2002). Pesticide indices are ratios between exposures, measured as pesticide concentration, and toxicity, given by some critical biological thresholds. In this way indices capture the effect of pesticide concentration on the ecosystem in a relatively simple manner. The difficulties are related to the existence of a large number of pesticides, each with different properties of significance to the environment. In this section we will present the Pesticide Toxicity Index (PTI) and discuss this index in relation to other alternative indices.

4.1 Construction of the Pesticide Toxicity Index

PTI is an index used in Sweden to evaluate the progress towards one of the 16 Swedish national environmental goals, namely a non-toxic environment. Different pesticides have different potential toxicity to environmental non-target organisms and through PTI their concentration found in surface water is related to the pesticides’ threshold value for the aquatic environment (Asp and Kreuger, 2004). The aim of using the PTI is to follow changes in water quality over time, to compare different regions and to evaluate the overall progress of national risk reduction programs.

PTI assesses the potential risk to the environment caused by pesticide use. When more substances active already at low concentrations are used, lower average concentration of pesticides in surface water does not necessarily indicate decreased risk to the aquatic environment. It is therefore of great importance to not only focus on the used amounts but also to see the actual concentrations of pesticides in water coming from agricultural land and the potential risks of damage to the environment caused by these substances. Thus PTI does not only focus on pesticide amounts in water but also on the possible harm this may do and the significance this can have on the environment. The importance of this has been stressed in previous literature. Falconer (2002) pointed out the possibility to use an indicator that reflects the pesticide amounts in the environment based on monitoring data. However, long-term, consistent, monitoring is costly and therefore scarce, which leaves the option of using well-established mathematical models. As part of the TEAMPEST project, we therefore intend to calculate concentration of pesticides through simulation models. In what follows, we first present how PTI is calculated and then how it can be used as an appropriate aquatic indicator together with the simulation model FOOTPRINT, recently developed within another EU-project.
The Pesticide Toxicity Index was developed by the U.S. Geological Survey (USGS) to assess the U.S. national environmental monitoring programme (National Water-Quality Assessment, NAWQA) (Munn et al., 2006). The PTI has been slightly adjusted to Swedish conditions in order to be used for following the work towards the national environmental goal, a non-toxic environment (Asp and Kreuger, 2005).

The index is calculated by dividing the concentration of a chemical compound found in for example a stream, with its threshold value. These toxicity ratios are then summed up for all substances which gives a relative toxicity caused by all pesticides (Equation 1). A high ratio indicates a high risk of negative effects to the aquatic environment caused by pesticides.

**Equation 1. Pesticide Toxicity Index**

\[
PTI = \sum_{i=1}^{n} \frac{E_i}{RV_i}
\]

- \(PTI\) = Pesticide Toxicity Index
- \(E_i\) = Concentration of pesticide \(i\)
- \(RV_i\) = Threshold value for pesticide \(i\)
- \(n\) = Number of pesticides used

In the USGS method, an index is being calculated for one taxonomic group relevant for the site of the sample. This then gives a median value calculated from the acute toxicity data concerning daphnia, invertebrates and/or fish. The Swedish index however, is calculated based on national threshold values developed by the Swedish Chemicals Inspectorate (KemI). In 2004 they developed threshold values for about 100 plant-protecting product and their decomposition products. Some of these were then modified in 2007 (KemI, 2008). Threshold values mentioned in this report are however not forensically binding and should therefore be considered as guideline values. The terminology “threshold value” will however be used throughout this paper.

The method used in Sweden for developing threshold values for surface waters is based on guidelines from the EU Water Framework Directive (2000/60/EEC) and the internationally accepted methods for risk assessment of chemicals gathered in the EU Technical Guidance Document (KemI, 2004).

A threshold value indicates at which level the substance potentially gets toxic to the aquatic environment, that is, the highest concentration when there are no negative effects in surface water due to the substance. The Swedish threshold values are a contexture of the toxicity to different aquatic organisms and are therefore meant to protect also the most sensitive organism in the system. This means that concentrations in surface water of a chemical compound above its threshold value forms a potential risk to the aquatic environment (Asp and Kreuger, 2005). The aquatic environment is considered to be more sensitive to pesticide exposure than the terrestrial environment. All organisms, including terrestrial, will therefore

---

3 The Technical Guidance Document (TGD) provides supplementary technical details on hazard identification, dose (concentration) – response (effect) assessment, exposure assessment and risk characterisation in relation to human health and the environment. The guidance is not legally binding.
be protected from potential risk if an index that shows at which level pesticides get toxic to aquatic organisms is used.

Swedish threshold values aim to protect organisms from long term effects due to exposure to pesticides and they are mainly calculated from studies made on chronic toxicity. The values do however not take into account contingent cumulative or synergistic effects. The values are annual averages and there are hence no guarantees that there will be no effects on some populations due to occasional higher emissions (Linderoth, 2008).

These threshold values are used today when calculating the PTI within the Swedish environmental monitoring programme. There are though some substances and decomposition products today or previously included in the programme for which Swedish threshold values are missing. This mostly concerns old substances, no longer allowed to use in Sweden. However, when calculating the index, all findings of all substances must be included. Therefore, Environmental Quality Standards (EQS) from the European Commission or threshold values developed in the Netherlands or Norway are used when Swedish threshold values are missing. Some of the substances found do however have threshold values below their analytical limit of detection. These substances, primarily a group of insecticides called pyrethroids, have therefore been excluded from the calculations of PTI for national indicator (Adielsson and Kreuger, 2008).

4.1.1 Use of the Pesticide Toxicity Index in Sweden today

PTI is used as a monitoring tool to assess the environmental risk from use of pesticides in Sweden. Concentrations of pesticides are measured in stream water from four catchment located in different agricultural regions of Sweden and PTI is calculated for each catchment. Water samples are collected 20-25 times per catchment and year.

Figures 4a and 4b below show the development of the PTI in the catchments from 2002 to 2008 (Adielsson et al. 2009). The national threshold values for pesticides were first presented by KemI in 2004, which initiated work on lowering analytical detection limits for the most toxic pesticides to match the level of the threshold values. As a result of this work a number of pesticides were monitored with substantially lower detection levels from 2005 and onwards and could also potentially be detected more frequently. This introduces an inconsistency in the PTI series, which was removed by excluding pesticides with analytical detection limits above its threshold value from the calculations (Figure 4a). This gives a slightly incomplete but a more consistent time-series of PTI including only those pesticides possible to detect at levels consistent with the threshold value. In Figure 4b calculations of PTI are presented including also pesticides that has not been analysed at low enough detection limits throughout the monitoring period. Thus improvements in detecting highly toxic pesticides, i.e. pesticides with low threshold values, may explain more irregularities and jumps in the last relative the early period of the monitoring period.

The four monitoring catchments represent different agricultural conditions, with differing soils, climate, crops and also pesticide use. This diversity might explain the current lack of an overall, consistent trend in PTI in the four areas. The catchment in Västergötaland, area O18, is relatively stable and does not show up- or downward trends. This catchment is dominated by relatively non-intensive, cereal growing agriculture and the potential risk from pesticide use seems to be lower, and at a more stable level, compared to the other three areas. This
catchment is located in a region with less pressure from fungicides and insecticides, with contributes to the lower level of pesticide use compared to the other three.

The PTI calculations in the catchment E21 in Östergötland show a completely different pattern. One can discern a weak upward trend between 2002 and 2008 in figure 4a but when pesticides with threshold values below detection limit are included the upward trend is even more evident (Figure 4b). There are several possible explanations, such as lowering attention to risk mitigation measures amongst the farmers in the area or the introduction of new, more toxic pesticides in the area. This latter explanation can be supported by more frequent findings of some highly toxic active ingredients in the stream, but still some peak concentrations are occurring at times when no run-off or leaching situation is at hand, thus supporting also the first explanation. Overall, there seems to be an upward trend in PTI based on monitoring data from catchment E21 during 2002-2008.

The third catchment in this monitoring program is catchment N34 in Halland. In Figure 4a we see that there is a weak increase in PTI but this trend is less clear in Figure 4b. When pesticides with threshold values less than the detection rate are included, PTI show large differences over the years. This is primarily due to the detection of esfenvalerat and deltametrin. In the last catchment monitored, M42 in Skåne, we see in Figure 4a that PTI is relatively stable between 2002 and 2005 with the exception of a small increase in 2003. The following two year there is a drop in PTI and in 2008 we a substantial increase in PTI. The increase in 2008 has to do with increased use of high risk pesticides such as MCPA and diflufenikan. In Figure 4b the detection of esfenvalerat and betacyflutrin increase PTI substantially.

4.1.2 Strengths and weaknesses of the Pesticide Toxicity Index

PTI is a good method to explain the effects pesticides have on the aquatic environment and the index has several benefits compared to other models and indices. It is relatively simple and transparent and therefore easy to communicate and to use. All findings of pesticides above their threshold values are considered to have negative effects on the aquatic environment and the index is, as pointed out before, sensitive to findings of pesticides above these threshold values. It therefore makes it possible to see changes in potential environmental risk over time and also to link these changes to different measures taken in order to reduce pesticide losses to water. Since all findings of all substances are included in the calculations, cases where concentration exceeds the toxic level will hence have an impact on the index and increase its value. There are other models, like for instance the HAIR model, which calculates risk to the aquatic environment caused by one active substance at a time. These calculations therefore end up with one risk index for each substance, PTI does instead calculate an aggregated risk index caused by all findings of all substances and the result is therefore presented in one single index. The index is however calculated as if the potential risk to the aquatic environment increases additive. An index value twice as big as an index calculated for another area does not necessarily mean that the potential risk to the environment is twice as big. There may be synergistic or additively effects of pesticides, not taken into account in these calculations which should be taken into consideration when reading the results.
Figure 4. Pesticide Toxicity Index for surface water for four catchments in Sweden.

a) excluding pesticides with threshold values below the analytical detection limit


b) including pesticides with threshold values below the analytical detection limit.


Notes: Pesticide Toxicity index are calculated based on monitoring data from the following four catchments in Sweden: O18: catchment in Västergötland; E21: catchment in Östergötland; N34: catchment in Halland; and M42 catchment in Skåne.
As for many other indicators, it is when balancing of scientific soundness and simplicity the difficulties appear. The strength of PTI compared to many other indices is that it is relatively simple and user friendly. The ambitions of the indicators REXTOX, SYSCOR and ADSCOR developed by the OECD (appendix 1) are limited by lack of required data and are difficult for policy makers and the wider public to understand and use (OECD, 2008). Also the EU funded HAIR project (appendix 2) face similar problems. Even though the indicators themselves are considered to be quite reliable the software lacks a complete database, transparency and user friendliness and more work will be put into the HAIR project in order to improve the model (van Hyfte et al., 2009). This is however not good enough for TEAMPEST since this project requires a model ready for use already today. The Water Quality Index (WQI) is an index used and developed in Canada (appendix 3) (CCME, 2001). Although it is flexible and adjustable to the needs and existing circumstances it is neither user-friendly nor easy to communicate. The simplicity of PTI is to a large extent due to the existing of threshold values at national and European level. The threshold values, which also are guiding values for policy making, summarizes the scientific knowledge and makes it more accessible. Thus knowledge and data availability are not as problematic for PTI as for the other indices mentioned above. A potential problem with PTI can however be the threshold values which may differ between the European countries. Substances included in the Water Framework Directive have environmental quality standards common for the entire EU. For other substances, threshold values are developed in each country separately.

Another possible problem with PTI is that it depends on the number of pesticides found in water which is related to the number of compounds and active ingredients approved for use in a specific country. To some extent this may capture the possibility that the use of a large number of pesticides increases synergies. However, PTI increases with the number of active ingredients found in an area and this may be an effect that makes comparison of indices for different countries difficult. A possible solution is to specifically take into account the number of active ingredients approved for use in a country together with the result.

5. Application of the Pesticide Toxicity Index

In the previous sections we saw that the Pesticide Toxicity Index is a ratio between concentrations of active compounds derived from pesticide use found in nature and their threshold values. Threshold values were discussed in the previous section, in this section we introduce the FOOTPRINT model for calculating concentrations and discuss and motivate using this model in the TEAMPEST program.

Simulation models are often used for calculating pesticide concentrations at higher aggregation levels. This is probably because there are no other realistic alternatives. For small catchments monitoring data may be used but at larger scales, such as regional, national or international levels, monitoring requires too many resources and is therefore unrealistic to do. Simulation has also the advantage to be flexible enough to allow analysis of many different circumstances with different economic, climatic and geographic conditions, which is interesting for policy. Simulated concentrations under different conditions are combined with the threshold values of specific pesticides to obtain information on the potential risk pesticides pose to the environment. In this project we are going to use FOOTPRINT as a method for
calculating pesticide concentrations. FOOTPRINT has been developed for aquatic environments and can accommodate the existing data on aquatic threshold values at national and EU levels. PTI, and hence the environmental risks from pesticides, can therefore be calculated in a consistent way by combining FOOTPRINT and the threshold values for aquatic environments, which is in line with the purposes of this report.

It is however important to be aware that the calculation of total exposure through any simulation model faces all the complications discussed in the previous sections. It is therefore important to test and validate the FOOTPRINT model. In this report we intend to use Swedish monitoring data from a catchment in Sweden for validation. In section 5.1 we present the simulation model FOOTPRINT and the Swedish monitoring program which will be used to validate and test the simulated data from FOOTPRINT.

### 5.1. FOOTPRINT

FOOTPRINT is a research project funded by the European Commission as part of the 6\textsuperscript{th} Framework Programme for Research and Technological Development (FP6). 15 partner institutions in nine EU countries are involved in the FOOTPRINT project which started in January 2006 and is planned to be finished in June 2009. The purpose of the project is to develop computer tools that can be used to evaluate and reduce risks of pesticides impact on water resources in the EU. The project aims to reach different users and is therefore developing three different tools which are being designed to operate at different scales. These are FOOT-NES, which focuses on the national and European scales, FOOT-CRS with a focus on catchment and regional scales and FOOT-FS which is on a farm scale. Although the tools work at different scales they share the same underlying structure and are based on the same scientific foundations. Each of these tools can be used to identify possible contamination pathways at different scales in the landscape and helps to identify which areas in the agricultural landscape that contribute the most to contamination of water resources. FOOTPRINT can also generate site-specific recommendations on how to reduce transport of pesticides to water and estimate the level of pesticides being transported to surface water and groundwater. (FOOTPRINT, 2008)

One of the major benefits with FOOTPRINT is its pesticide database which, compared to other databases such as the HAIR database, can be considered to be very transparent. Here the best pesticide data available are brought together in on single dataset which is updated on a regular basis. The database has been developed by the Agriculture & Environment Research Unit (AERU) at the University of Hertfordshire as a part of the FOOTPRINT project and contains a lot of information regarding environmental fate and ecotoxicological properties for a large number of pesticides as well as their metabolites. The database is well documented and transparent as information for each pesticide can be traced back to the sources. This database is probably the most extended one there is to find today and it has, according to the FOOTPRINT webpage, already become a reference in Europe as well as the rest of the world. (FOOTPRINT, 2008)

The database contains data regarding all pesticides and selected metabolites found in Annex I of Regulation (EC) No 396/2005. About 650 active substances and 200 metabolites are comprised in the list. There are two kinds of information found in the database: chemical and physical properties and ecotoxicological data. Additional information like alternative chemical names and a list of which EU member states where the active substances are registered can also be found.
The best available information about pesticide properties are from papers produced as part of the EU review process and information found in these documents forms an important part of the FOOTPRINT database. When EU documents are missing, information from other references have been used instead. This information has for instance been collected from various national government departments, different research projects and manufacturers’ safety datasheets. Each data item is tagged with a code in order to show the origin of the data. The tag also includes a quality score, ranking from 1 (low) to 5 (high), which represents the developer’s faith in the quality of the data.

The FOOTPRINT models will be used in WP3 to estimate exposure of pesticides to surface water. Using these models makes it possible to build different scenarios by altering different farm management practices, climate and geographical conditions to calculate exposure to pesticides. The FOOTPRINT project is ongoing and the three tools are not yet fully developed and therefore not ready to be used by the general public. However, they will be made available in the public domain at the end of the project (June 2009). A test version of the FOOT-FS model is currently in circulation and it is possible to use it among FOOTPRINT partners. Although the FOOTPRINT tools are not yet fully developed we chose to use these models because we are convinced that they best fit the objectives of the TEAMPEST program. Although the best results may not be achieved before the FOOTPRINT tools are fully developed in June 2009, FOOTPRINT has the potential to give the best estimates of pesticide exposure.

5.2 Validation of FOOTPRINT

In general, the process of validation is important for developing good indicators. A monitoring programme in a catchment in Östergötland, located in the southern part of Sweden, measures pesticide concentrations in surface waters and can be used to test and evaluate the simulated concentrations from the FOOTPRINT model. For this purpose we intend to run the FOOTPRINT model with the information known for the catchment in Östergötland (such as crops, management practices and pesticide management) and then compare the modelled pesticide concentrations with the ones measured in reality in the catchment. This kind of test against real data has yet not been done for the FOOTPRINT model.

5.2.1 Swedish national monitoring programme

Within the Swedish national monitoring programme, done by the Swedish University of Agricultural Sciences (SLU) on behalf of the Swedish Environmental Protection Agency, water samples are collected on a regularly basis in four monitoring catchments. In 1990 water sampling started in catchment M 42 (Skåne). This sampling was in 2002 enlarged to also include the three catchments N 34 (Halland), E 21 (Östergötland) and O 18 (Västergötland). These areas, so called type areas, represent bigger geographical areas with different weather conditions, soil types and cultivation patterns in some of the most agricultural dominated locations in Sweden (Figure 5). In these catchments, sampling of surface water, sediment and shallow groundwater are done automatically on a weekly basis from May until October. Samples from surface water are taken automatically every 80th minute and are then collected on a weekly basis. The concentration in one sample does thereby represent the average concentration for one week. The water samples are then analyzed for about 90 different
substances (herbicides, insecticides and fungicides) that are commonly used, prone for leaking, have low threshold values and/or are included in the Water Framework Directive (2000/60/EG) as a prioritized substance. The results of the environmental monitoring programme are published annually by SLU. According to the most recent report, regarding year 2007, most of the substances analyzed for are not found in any of the samples taken. If however they are detected, the concentration in the sample in most cases is found under its threshold value. In 2007, 30-45 % of the analyzed substances were found in the surface water samples and 19 substances were found in concentrations above its threshold value. Most of the detections above the threshold values were from the herbicides diflufenican and MCPA. A good knowledge of background information like pesticide use, water flow and precipitation in these areas, combined with these analyses of regularly taken water samples thereby gives a good documentation of the state of the environment as well as the trends. (Adielsson and Kreuger, 2008)

Owing to annual interviews with farmers operating in these catchments, the monitoring programme also has a lot of information on crops, fertilization and pesticide use (type of pesticide, dosage and time) in these areas. This kind of information provides a possibility to link management practices and pesticide use to occurrence in the aquatic environment.

Catchment E 21 is located in the county of Östergötland, one of the most intensively cultivated regions in Sweden. The catchment comprises about 1 700 hectares, of which 89 % is arable land. The dominating soil type is loam. Water sampling started here in spring 2002 and is since then taken during May-October with about 20 samples per year. The agricultural practices are within this catchment dominated by cereal crops. Other crops cultivated in the area are oil plants, potatoes, peas and grass ley. (Kreuger and Adielsson, 2008)
5.3 Integrating FOOTPRINT with PTI

Since FOOTPRINT only calculates concentrations of substances in water, a combination of the model and the Pesticide Toxicity Index is a good way to take potential risks to the environment into consideration. Once the model has been tested against real data, an up-scaling to bigger areas will be possible to do. We will then also test different measures in the model in order to see how effective they are in decreasing the index value. It can therefore, by using the FOOTPRINT model in combination with PTI, be possible to push agricultural development towards a use of less toxic substances.

Monitoring data are costly and the frequency of monitoring is often not high enough to capture all pesticide concentration peaks. Monitoring data may therefore miss acute toxicity causing environmental problems. To test a model against monitoring data may therefore be somewhat fallacious. A second problem is that there may be different explanations to the differences between simulated and monitored pesticide concentrations. If these differences in data are related to some properties in the monitored area then testing the model may lead to wrong estimations in other areas.

Most of the threshold values used for calculating PTI today is developed in Sweden and these threshold values are not necessarily the same as the ones used in other European countries. It can therefore, in order to make the calculations more adjusted to the rest of EU, be recommended to use Environmental Quality Standards developed by the European Commission. The toxicology to aquatic invertebrates are also stated in the FOOTPRINT database which then also might be used in order to adjust the calculations to European situations.

6. Lessons learned

The process of designing environmental indicators represents a trade off between accurately describing potential risks to the environment and simplicity. The more accurate an indicator is the more information is required and processing this information often makes the indicator more complicated and difficult to use. Choosing an appropriate indicator depends on existing circumstances, the complexity of the environmental problem and the intended use of the indicator. Indicators for complicated environmental problems are much more challenging to design since these cases often involve a higher degree of scientific uncertainty with respect to factors determining the risks to the environment. In these cases, designing an indicator is very difficult and the validity of -the chosen indicator will inevitably be questioned due to scientific controversies. With or without these uncertainties the amount of information incorporated into an indicator may still be large. If the indicator is to be used at high aggregation levels that include many ecosystems, climatic regions, number of sources and dispersion of emitters then this indicator must also accommodate a great deal of information.

Design of environmental indicators for pesticides is very challenging. Pesticides are used to manage many different types of pests in many different environments. This implies that many different pesticides with different characteristics are spread over large geographical areas. Pesticide indicators therefore need a great deal of information to capture the risk of pesticide use on the environment. Perhaps the most challenging step when building an indicator is the aggregation of environmental risks posed by different active ingredients on many ecosystems under different climatic and geographic circumstances. In the past, many indicators have
simplified the problem by basing information on the quantity of pesticides applied. These may be good indicators for estimating the quantities of pesticides in the environment but they do not provide a sufficient explanation of the potential risks to the environment. Over the past two decades, pesticide indicators have been developed with the specific ambition of providing information on environmental risks, for example, through the use of indices.

The PTI developed for policy purposes, is both simple and user-friendly. The calculations for compiling the index are based on all findings of all substances irrespective of whether the concentrations exceed threshold values or not. The PTI relates actual concentrations to their threshold values. This aspect is of great importance since different substances are associated with different degree of toxicity to the environment. Measured concentrations of all substances contribute proportionally with their associated potential risk to the index value. Pesticides with a high toxicity have a high impact on the index value even if the concentration compared to other pesticides is low. Threshold values capture the potential risk of each substance to the aquatic environment and preclude the need for additional weighting.

With respect to the complexity of the potential risks to the aquatic environment from pesticide use, the PTI represents a reasonable balance between scientific accurateness and simplicity. The index uses data on pesticide concentrations to capture environmental risks in a relatively simple and user-friendly way. The simplicity of the index also makes it easy to communicate with, which is a useful property when policy makers present environmental policy to the general public.

The PTI is a sum of the ratios between pesticide concentrations and threshold values which makes the index very sensitive to the number of pesticides found. However, there may also be synergy effects between compounds since there are many active ingredients and many pesticides formulations in use. The risk for these types of effects increases with an increase in the amount of pesticides used and found in surface waters. In addition, potential synergy effects may also differ among EU members since the number of pesticides allowed for use in each member state also varies. The PTI in its present form is not able to take into account any of these potential synergy effects.

In the upcoming TEAMPEST tasks (in WP3, WP5, WP6 and WP8), the FOOTPRINT model will be used to calculate concentrations of pesticide compounds in surface water. These modelled concentrations will serve as input to the PTI to estimate the total environmental risk to the aquatic environment. The flexibility of the FOOTPRINT simulation model allows policy analysis to be performed under different circumstances and will be used to identify the most efficient policies to decrease the environmental risks from pesticide use. The FOOTPRINT model is EU funded and has a very transparent database which makes it easy to use and understand. The database includes most pesticides today used within the EU as well as their properties. Sources of the information in the database are clearly described and updated on a regularly basis. In contrast to the HAIR model, the FOOTPRINT was explicitly developed for modelling pesticide use and therefore better suits the needs of the TEAMPEST project.

Finally, guidelines for indicator selection developed by the European Environmental Agency (EEA) are considered to be of relevance for the TEAMPEST project. The EEA criteria are practical and policy oriented and in line with the objectives of TEAMPEST. According to the EEA criteria a good indicator should be policy relevant and possible to use in order to see
progress toward environmental objectives. The PTI is already used in Sweden to follow up on work towards one of the national environmental goals and is also easy to understand.

It is the conclusion of the authors of this report that:

- the PTI is a good pesticide environmental index given the complexity of pesticide environmental risks and the availability of data and
- the advantages of PTI are best utilized in combination with the FOOTPRINT model, which was specifically developed for assessing pesticide concentrations in water.
- the PTI can be used to analyze the impact on the environment of alternative policy scenarios to reduce the external costs of pesticide use.
References


Data sources


Appendix 1. Technical description of REXTOX, ADSCOR and SYSCOR

Within the OECD pesticide programme an aquatic risk indicator project started in 1998 with the purpose to develop indicators that could track trends in risk resulting from agricultural pesticide use. The project terminated in 2001 and developed and tested three different models that could be used to derive pesticide risk indicators – REXTOX, ADSCOR and SYSCOR. These indicators all roughly estimate how much of an applied pesticide that migrates from the field to surface waters and also its risk to aquatic organisms. The models share a similar underlying structure and they do also deal with toxicity in the same way, using short-term or long-term values for fish, daphnia or algae. These separate risk indices for each of the three aquatic organisms are then added together in order to obtain one risk index for aquatic risk in general. They however somewhere in the way they estimate exposure and how they weight different variables. (OECD, 2000)

REXTOX (Ratio of EXposure to TOXicity) uses simplified mathematical models of the mechanisms of pesticide fate and movement in the environment to estimate pesticide concentrations in surface water as a function of use. These concentrations are then multiplied by the total amount of pesticides used in order to estimate the exposure which are linked to hazard data to estimate risk. These exposure and risk values are then finally combined for all uses of all pesticides which provide the final aggregated indicator.

ADSCOR (ADditive SCORing) converts true values of pesticide use parameters to scores reflecting its general risk contribution. These scores are added together to obtain an exposure score for each use which then are summed up into an aggregated exposure score for each pesticide. Finally, like in REXTOX, this score is linked to hazard data, and exposure and risk values are combined.

SYSCOR (SYnergistic SCORing) links the variables in a hierarchical way in order to reflect both their importance and their interaction. This linking generates scores for the different variables which then are added together.

**Table 1. Structure of OECD Aquatic Risk Indicators**

<table>
<thead>
<tr>
<th></th>
<th>REXTOX</th>
<th>ADSCOR</th>
<th>SYSCOR</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Uses true values for all variables</td>
<td>Uses true values for area and toxicity</td>
<td>Uses true values for toxicity and scored values for all other variables</td>
</tr>
<tr>
<td></td>
<td>Links variables in mechanistic models</td>
<td>Uses scored values for other exposure variables</td>
<td>Always scaled</td>
</tr>
<tr>
<td></td>
<td>Calculates either scaled or unscaled</td>
<td>Links exposure scores by addition</td>
<td>Links scores logically with synergistic effect</td>
</tr>
</tbody>
</table>

Source: OECD, 2002
Appendix 2. Technical description of HAIR

The HAIR project (HArmonised environmental Indicators for pesticide Risk) started in 2003 and finished 2007 and was financed under the EU’s 6th research framework programme. Its purpose was to develop a set of harmonised indicators that could indicate an overall risk of pesticides to the environment and to human health. Finally, about 50 indicators were developed, covering the aquatic environment, groundwater, the terrestrial environment, consumers and operators/bystanders. All of these indicators do however not appear as an output in the final software. (van Hyfte et al., 2008)

In the aquatic risk indicator developed within this project, the final risk assessment is expressed as the Exposure to Toxicity Ratio \( (ETR) \). Two separate ratios are calculated, one for the acute risk and the other regarding chronic risk. The potential acute biological risk is based on standard tests and is calculated as the ratio of short-term exposure \( (sPEC) \) to the lethal concentrations \( (LC50) \) for the exposed organisms algae, daphnia and fish. It is calculated as follows:

\[
ETR_{\text{acute}} = \frac{sPEC}{LC50_{\text{species}}}
\]

The potential chronic biological risk is calculated in a similar way, as the ratio of long-term exposure \( (lPEC) \) to the no-effect concentrations \( (NOEC) \):

\[
ETR_{\text{chronic}} = \frac{lPEC}{NOEC_{\text{species}}}
\]

The ratio calculations are done for each active substance separately and one for each exposed organism (algae, daphnia and fish). The potential risk is not summed up to one value and there are hence many values to take into account when evaluating the total potential environmental risk caused by the use of pesticides (Strassemeyer et al., 2007).

The indicator outputs are presented at three different scales. In Case 1 pesticide data are available on field level and this therefore forms the best case of data availability. Pesticide application patterns are in Case 2 available on regional basis and pesticide use data as a set of crop related application patterns. In the final case, Case 3, application conditions are available on a regional basis and pesticide use data as volumes of active ingredient applied per crop.

The HAIR database is made up of datasets obtained from the project partners and the EU Environmental Quality Standards are therefore not included. The database contains mostly numerical information on active ingredients and is designed to function as an internal data source for calculations within the project. Data can be found concerning chemical identity, physico-chemical parameters, environmental fate, toxicity to various organisms and some parameters regarding toxicity to humans. The fifty percent lethal concentrations \( (LC50) \) and the no-effect concentrations \( (NOEC) \) for each active substance are calculated for the exposed organisms algae, daphnia and fish. (van Vlaardingen et al., 2007)
Appendix 3. Technical description of WQI

The Water Quality Index, (WQI), was developed by the Water Quality Index Technical Subcommittee under the Canadian Council of Ministers of the Environment, (CCME). This index is calculated by combining monitoring data regarding the amount of substances for which the concentration found in water is exceeding its threshold value with how much and how often this happens for each single substance. The threshold values used when calculating this index are the Canadian Water Quality Guidelines (WQG) which, according to the definition, aims to protect all kinds of aquatic life. (CCME, 2001)

The extent of the index regarding water bodies (e.g. rivers or lakes) and time period can be varied and therefore decided by each single user. The user can furthermore decide which and how many other variables concerning different quality aspects (e.g. metals, pH, total-P and bacteria) that also should be included in the index. The method is therefore considered to be quite flexible. The possible options do, on the other hand, also increase the risk of subjective selection of variables and makes comparison of indices calculated for different regions more complicated.

The calculation of the index is a mathematical combination of three different factors that relate to water quality objectives: the number of substances for which threshold values are exceeded ($F_1$), the frequency with which the threshold values are exceeded ($F_2$) and the sum of all concentrations of substances exceeding their threshold values ($F_3$). The Water Quality Index is then finally calculated by summarizing the three factors as vectors and dividing this with 1,732 which normalizes every factor to a value between 0 and 100.

Equation 4. Water Quality Index

$$WQI = 100 - \left( \frac{F_1^2 + F_2^2 + F_3^2}{1.732} \right)$$

In Canada the index value is used in a ranking system where 0 represents very bad water quality and 100 corresponds to an excellent quality. The ranking system, shown in table A2, is made up of five different categories; excellent, good, fair, marginal or poor.

Table 2. Classification of water quality according to the Canadian Water Quality Index

<table>
<thead>
<tr>
<th>Quality</th>
<th>Index value</th>
<th>Water quality</th>
<th>Water conditions</th>
</tr>
</thead>
<tbody>
<tr>
<td>Excellent</td>
<td>95 – 100</td>
<td>Absence of threat or impairment</td>
<td>Good and very close to natural levels</td>
</tr>
<tr>
<td>Good</td>
<td>80 – 94</td>
<td>Minor degree of threat or impairment</td>
<td>Close to natural and desirable levels</td>
</tr>
<tr>
<td>Fair</td>
<td>65 – 79</td>
<td>Occasionally threat or impairment</td>
<td>Sometimes departed from natural or desirable levels</td>
</tr>
<tr>
<td>Marginal</td>
<td>45 – 65</td>
<td>Frequently threat or impairment</td>
<td>Often departed from natural and desirable levels</td>
</tr>
<tr>
<td>Poor</td>
<td>0 – 44</td>
<td>Almost always threat or impairment</td>
<td>Usually departed from natural and desirable levels</td>
</tr>
</tbody>
</table>

Source: CCME, 2001