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National Agri-Environmental Standards Initiative (NAESI)

Report No. 1-17

Development of Comparative Environmental Risk Assessment Tools for Pesticides in Support of Standard Development At Environment Canada

Technical Series 2005

Photos:
Bottom Left- clockwise

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Crop spraying: Corel CD photo # 95C2840
Elk Creek, BC: Joseph Culp, National Water Research Institute, Environment Canada
Prairie smoke and bee: Emily Wallace, Prairie Northern Region, Environment Canada

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DEVELOPMENT OF COMPARATIVE ENVIRONMENTAL RISK ASSESSMENT TOOLS FOR PESTICIDES IN SUPPORT OF STANDARD DEVELOPMENT AT ENVIRONMENT CANADA

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NOTE TO READERS

The National Agri-Environmental Standards Initiative (NAESI) is a four-year (2004-2008) project between Environment Canada (EC) and Agriculture and Agri-Food Canada (AAFC) and is one of many initiatives under AAFC's Agriculture Policy Framework (APF). The goals of the National Agri-Environmental Standards Initiative include:

- Establishing non-regulatory national environmental performance standards (with regional application) that support common EC and AAFC goals for the environment
- Evaluating standards attainable by environmentally-beneficial agricultural production and management practices; and
- Increasing understanding of relationships between agriculture and the environment.

Under NAESI, agri-environmental performance standards (i.e., outcome-based standards) will be established that identify both desired levels of environmental condition and levels considered achievable based on available technology and practice. These standards will be integrated by AAFC into beneficial agricultural management systems and practices to help reduce environmental risks. Additionally, these will provide benefits to the health and supply of water, health of soils, health of air and the atmosphere; and ensure compatibility between biodiversity and agriculture. Standards are being developed in four thematic areas: Air, Biodiversity, Pesticides, and Water. Outcomes from NAESI will contribute to the APF goals of improved stewardship by agricultural producers of land, water, air and biodiversity and increased Canadian and international confidence that food from the Canadian agriculture and food sector is being produced in a safe and environmentally sound manner.

The development of agri-environmental performance standard involves science-based assessments of relative risk and the determination of desired environmental quality. As such, the National Agri-Environmental Standards Initiative (NAESI) Technical Series is dedicated to the consolidation and dissemination of the scientific knowledge, information, and tools produced through this program that will be used by Environment Canada as the scientific basis for the development and delivery of environmental performance standards. Reports in the Technical Series are available in the language (English or French) in which they were originally prepared and represent theme-specific deliverables. As the intention of this series is to provide an easily navigable and consolidated means of reporting on NAESI's yearly activities and progress, the detailed findings summarized in this series may, in fact, be published elsewhere, for example, as scientific papers in peer-reviewed journals.

This report provides scientific information to partially fulfill deliverables under the Pesticide theme of NAESI. This report was written by P. Mineau and M. Whiteside, National Wildlife Research Center, Environment Canada. The report was edited and formatted by Denise Davy to meet the criteria of the NAESI Technical Series. The information in this document is current as of when the document was originally prepared. For additional information regarding this publication, please contact:

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NOTE À L'INTENTION DES LECTEURS

L'Initiative nationale d'élaboration de normes agroenvironnementales (INENA) est un projet de quatre ans (2004-2008) mené conjointement par Environnement Canada (EC) et Agriculture et Agroalimentaire Canada (AAC) et l'une des nombreuses initiatives qui s'inscrit dans le Cadre stratégique pour l'agriculture (CSA) d'AAC. Elle a notamment comme objectifs :

d'établir des normes nationales de rendement environnemental non réglementaires (applicables dans les régions) qui soutiennent les objectifs communs d'EC et d'AAC en ce qui concerne l'environnement;

d'évaluer des normes qui sont réalisables par des pratiques de production et de gestion agricoles avantageuses pour l'environnement;

de faire mieux comprendre les liens entre l'agriculture et l'environnement.

Dans le cadre de l'INENA, des normes de rendement agroenvironnementales (c.-à-d. des normes axées sur les résultats) seront établies pour déterminer les niveaux de qualité environnementale souhaités et les niveaux considérés comme réalisables au moyen des meilleures technologies et pratiques disponibles. AAC intégrera ces normes dans des systèmes et pratiques de gestion bénéfiques en agriculture afin d'aider à réduire les risques pour l'environnement. De plus, elles amélioreront l'approvisionnement en eau et la qualité de celle-ci, la qualité des sols et celle de l'air et de l'atmosphère, et assureront la compatibilité entre la biodiversité et l'agriculture. Des normes sont en voie d'être élaborées dans quatre domaines thématiques : l'air, la biodiversité, les pesticides et l'eau. Les résultats de l'INENA contribueront aux objectifs du CSA, soit d'améliorer la gérance des terres, de l'eau, de l'air et de la biodiversité par les producteurs agricoles et d'accroître la confiance du Canada et d'autres pays dans le fait que les aliments produits par les agriculteurs et le secteur de l'alimentation du Canada le sont d'une manière sécuritaire et soucieuse de l'environnement.

L'élaboration de normes de rendement agroenvironnementales comporte des évaluations scientifiques des risques relatifs et la détermination de la qualité environnementale souhaitée. Comme telle, la Série technique de l'INENA vise à regrouper et diffuser les connaissances, les informations et les outils scientifiques qui sont produits grâce à ce programme et dont Environnement Canada se servira comme fondement scientifique afin d'élaborer et de transmettre des normes de rendement environnemental. Les rapports compris dans la Série technique sont disponibles dans la langue (français ou anglais) dans laquelle ils ont été rédigés au départ et constituent des réalisations attendues propres à un thème en particulier. Comme cette série a pour objectif de fournir un moyen intégré et facile à consulter de faire rapport sur les activités et les progrès réalisés durant l'année dans le cadre de l'INENA, les conclusions détaillées qui sont résumées dans la série peuvent, en fait, être publiées ailleurs comme sous forme d'articles scientifiques de journaux soumis à l'évaluation par les pairs.

Le présent rapport fournit des données scientifiques afin de produire en partie les réalisations attendues pour le thème des pesticides dans le cadre de l'INENA. Ca rapport a été rédigé par P. Mineau et M. Whiteside du Centre national de la recherché faunique d'Environnement Canada. De plus, il a été révisé et formaté par Denise Davy selon les critères établis pour la Série technique de l'INENA. L'information contenue dans ce document était à jour au moment de sa rédaction. Pour plus de renseignements sur cette publication, veuillez communiquer avec l'organisme suivant :

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INTRODUCTION

Overall Research Objectives

Environment Canada has been tasked with developing environmental standards for implementation in Agriculture and Agri-Food Canada's Agricultural Policy Framework (AAFC; APF). Setting a 'standard' of acceptability for pesticide use, whether ideal and achievable or desirable but perhaps premature (Caux and Jiapizian, 2004) implies that we can objectively measure the impact that pesticides are having on key environmental components. Whereas it may be possible to 'cherry-pick' a few absolute standards of good agricultural practice (e.g. no pesticide use should lead to a fish or bird kill), many pesticide impacts may be much more subtle or graded in nature. By their nature, pesticides carry an inherently high risk to some segment of the environment and choosing the right product often becomes a question of trading off a high risk in one environmental component for a high risk in another. Water quality guidelines are workable standards because it is possible to measure the concentration of the chemicals in question in water to check for exceeded levels in the environment. However, not all pesticides are equally well covered by water sampling nor are all environmental impacts mediated through movement of the chemical into water.

The National Wildlife Research Center (NWRC) was tasked with developing comparative environmental risk assessment tools for pesticides in support of standard development. The development of standardised pesticide assessment tools will enable EC to:

Prioritise in-use pesticides for the development of Water Quality Guidelines

Provide environmentally-oriented advice to AAFC under the APF, allowing for the promotion of reduced risk pest management strategies

Objectively assess the environmental impact of alternative pesticide products and prioritize them for research and monitoring

Elsewhere, pesticide risk assessment measurement systems have also been used to develop agricultural certification systems in order to reassure customers that their food is being grown with the utmost care for the environment (e.g. 'Protected Harvest': http://protectedharvest.org) see introduction to Appendix B). This is another ultimate goal of the APF that outputs from this project should be able to contribute to.

Objective of the Current Milestone

This report is the first milestone in this multi-year project. It is intended as a scoping exercise and thought starter and provides a critical review of existing risk assessment systems. We have isolated a number of evaluation systems which assess valued components of the environment, namely terrestrial wildlife (birds, mammals, amphibians and reptiles), beneficial insects (pollinators, predators, parasitoids), soil micro and mega-fauna, aquatic invertebrates and fish as well as terrestrial and aquatic non-target plants. Some twenty-nine systems described in the published or gray literature from 1992 to current (Table 1) were retained for analysis. A few others were identified but have not yet been fully examined pending receipt of reference material. Several of the evaluation systems we encountered included human health components (Table 2), but only the environmental components are discussed here given the scope of the NAESI project. We focused primarily on toxicity-based risk assessment systems as opposed to measures of environmental distribution. Risk of movement into groundwater, for instance, is the main component of another initiative under the APF (Alan Cessna & Ross McQueen, pers. comm.).

However, most risk assessment systems need to consider how exposures will be predicted and this will be discussed here also, albeit briefly. The questions addressed in this report often relate to the choice of input variables and how data are manipulated as they are incorporated into equations in order to derive some sort of index. Our intent is not to re-invent the wheel where it isn't necessary to do so but to propose the most scientifically sound (but practicable) toxicity-based risk assessment tool for each component of the ecosystem and assess to what extent the proposed tool has proven itself or been properly validated. In later phase of this project, we intend to explore what validation steps are still required and carry out some of the needed analyses.

As a case study, we will propose a risk assessment system for acute bird impacts (see Appendix B). This proposal has already had the benefit of a commenting round from a group of international avian toxicity experts from universities, government and industry. We present it here as an example of the level of validation and confidence we would like to reach with every sub-component of a comprehensive environmental risk assessment system.

Pesticide Risk Assessment Systems vs. indicators

Many pesticide risk assessment tools have been developed throughout the world over the past decade. Depending on the intended use of the resulting metrics, they have been variously called hazard rankings, yardsticks, indicators, screening benchmarks, relative risk rankings, risk assessment tools … and more. Levitan ([2000]) proposed a typology which distinguishes between risk assessment tools depending on whether they are intended for grower decision support systems, for 'eco-labelling' purposes or to provide governments and others with an estimate of ecological damage. She distinguished between 'indicators' and 'impact assessment systems'.

‘Indicators’ tend to be succinct summaries and integrations of various trends or highlights which evolved within a framework of policy analysis and risk communication. ‘Impact assessment systems’, on the other hand, retain the ecosystemic perspective as well as the depth and complexity which are appropriate for the level of knowledge of a particular environmental component. Levitan’s typology has not met with uniform acceptance and, therefore, most of the European literature for example refers to ‘indicators’ regardless of the structure or intended function of the calculated risk index. Risk assessment systems can provide insight not always available from evaluations carried out by pesticide registration authorities. The latter typically consider pesticides singly (product-specific registration procedures) and make registration decisions often under imprecise concepts of risk and benefit.

Existing reviews of Pesticide Risk Assessment Systems

For our purposes, the most useful recent comparison of available risk assessment systems is that of Reus et al. (1999, 2002) under the European CAPER project. The intent was to identify and compare management and information tools that primarily provide information to farmers and applicators on the absolute and relative risks of pesticides to the environment in order to tie in to standards of compliance and Good Agricultural Practices. This is a goal very similar to ours. Reus and colleagues looked at 8 different European systems, namely the Dutch Environmental Yardstick (EYP), Danish Hasse Diagram, German SYNOPS.2, United Kingdom’s p-EMA, France’s Ipest, the Italian EPRIP, the Belgian SyPEP and Swedish PERI. The Norwegian model (one that has been extensively discussed by OECD task groups) was not included in the comparison. The authors produced a number of typical pesticide use scenarios and compared the relative agreement between the systems. Relative rankings did vary considerably depending on the chosen system although a good part of this variation came from the integration of the various sub-measurements (i.e. individual indices). All assessment systems considered the risks to aquatic life, seven considered ground water contamination and five the risk to soil organisms – essentially earthworms. None included terrestrial vertebrates, epygeal invertebrates or indirect effects. Most alarming, all are severely limited in not being to adequately compare risk among different formulation types such as granulars and seed treatments although some correction factors were proposed in the Dutch EYP. Another useful review was that of Van Bol et al. (2002), part of Belgium's contribution to the OECD indicator project.

Following a meeting of experts, the CAPER project reached a number of conclusions about the ideal instruments from the point of view of providing advice on product choice:

They favoured those instruments that included application-specific information such as method and timing of application, formulation and site characteristics. Despite the higher inherent difficulty of these more detailed assessment tools, we concur with this conclusion. It is imperative to move as much as possible from a measure of hazard to one of actual risk. As a crude demonstration of the difference, a pesticide of high inherent aquatic hazard has a negligible aquatic risk when used a safe distance from water. Because our assessment system is to cover the whole of Canada, we need to allow application specifics to alter the relative ranking of the alternative pesticides (tradeoffs). However, Reus and colleagues also found that the relative ranking of products did not change very much in light of different environmental conditions (wind speed, slope, soil type etc…). The exception was proximity to water bodies because it changed the emphasis from drift to runoff, with the latter but not the former dependant on the different physicochemical properties of the pesticide. This suggests that, as a first step, it might be sufficient to identify a few key exposure scenarios (drift or overspray vs. maximal runoff) to see how much this changes the risk assessment.

They favoured keeping individual scores as much as possible and the ideal system would have to be amenable to integration into a farm-specific decision-support system. We agree with this conclusion. Some of the systems such as SYNOPS have devised a visual multidimensional method of presenting the results on the different environmental segments.

They expressed concerns about the relationship between risk assessment systems and regulatory evaluation procedures. Ideally, there should be good agreement between the two – at least for the components of the environment that are assessed by both. In theory, we do not see the need for there to be any difference between a relative risk ranking system and a regulatory evaluation system. The only significant difference should be in the output: the former should provide one with a score which can be ranked or compared to a standard of best farming practice – the latter need only assess whether the product meets certain cost/benefit criteria. In practice however, regulatory evaluation systems tend to be codified and therefore very slow in changing. Therefore, they are perpetually lagging behind the available science.

They highlighted the need to include non-pesticide environmental damage in the assessment. For example, p-EMA is part of a larger EMA system which considers other farm-based management strategies. Similarly for the Swedish PERI which is one of a suite of assessment tools. Although we also agree in principle to the inclusion of non-pesticide impacts, especially when comparing pesticidal and non-pesticidal alternatives, this falls outside our current mandate.

They stressed the need for validation but then concluded that, in the absence of true validation of field effects, expert judgment would have to suffice. The lack of validated outputs is one of the most serious problems with either regulatory risk assessments or risk ranking tools. We believe that this is a priority area of research for all Departments having a stake in pesticides.

They stressed the problems associated with incomplete data sets. Clearly, the more complicated the system used, the more this becomes an issue. It has been widely recognized that the best assessment tools represent a compromise in that respect.

Four of the systems used a risk ratio approach to the evaluation of risk (EYP, SYNOPS.2, SyPEP and EPRIP). Because this is analogous to regulatory assessments, we believe these methods are the most suitable as possible departure points. Also, some of the other point-based systems such as Ipest and p-EMA placed a great deal of emphasis on total pesticide volumes. As will be discussed later, we favour a system that assesses the risk of individual applications and, as a last step only, adds up the number of treated hectares. The SyPEP system is different in that it relies on the existence of existing water quality criteria and looks for predicted exceedence. Because criteria are not set for all pesticides, we do not consider this to be practicable for our purposes. Reus and colleagues concluded that no one system was ideal and there was a need to develop a harmonized European system. This exercise (HAIR) is currently underway. The authors of the present report have been invited to attend meetings of the pan-European HAIR initiative as observers. Given the considerable sums being spent on that exercise, we intend to track their progress over the next few years and make use of any results that prove applicable to the NAESI project.

Another review exercise of note was conducted by the OECD between 1998 and 2001. The goal of that exercise was to provide OECD countries with the ideal tool with which to track aggregated risk and risk reduction targets – clearly a ‘higher level’ purpose than envisioned by the participants of the CAPER review. Hence, in the typology referred to earlier, the goal was to find the optimal policy ‘indicator’ rather than a more complex risk assessment system. For the purpose of the first part of the exercise (ARI), only aquatic risk was considered. (Another working group is considering terrestrial indicators but a report is not yet available.) The working group looked at SYNOPS (investigated by the CAPER group) but also considered the Danish Load and Application Indices, and the aquatic portion of the Norwegian Indicator (NARI). More importantly for their purpose, they created three very different indicators, based on different ways of scoring and aggregating use data, fate variables, application site variables and hazard (toxicity) data. They concluded that their different indicators could produce radically different risk trends over time but, in the absence of validation, were hard-pressed to decide which representation was most correct. At issue was the relative importance of some of the application site variables, notably the extent to which buffer zones mandated on the labels would or would not be followed.

Several of the principal conclusions of the OECD exercise were similar to those reached by the CAPER exercise:

Indicators should be designed with a specific purpose in mind

To be most useful, indicators should be calculated at low levels of integration

Indicators allowing for work at different spatial scales are best

Data are often incomplete or non-existent and this is a large source of uncertainty for indicators. Chronic toxicity data are often missing and the participants concluded that there was little point in developing long-term (chronic) indicators until that gap was filled.

Validation of indicators is highly desirable but difficult.

Indicators are most useful in showing trends because the actual values produced are often meaningless.

Indicators should be consistent with regulatory risk assessments

Relatively simple aggregate indicators such as the Danish Frequency of Application Index and Load Index as well as the Norwegian Indicator, while potentially adequate for policy use, were inadequate for more narrowly focused assessments such as a comparison of alternative pest management practices.

The OECD review did consider the choice of toxicity data e.g. “worst case”, “best case” or “most likely” and concluded that, for a simple index such as the Danish load index, the exact choice mattered little as long as the choice was consistent over time. Closer examination of the time trends, however, reveals a number of crossovers. We believe the importance of choosing the right toxicity endpoint will increase in the more narrowly focused rankings of pesticide options.

GENERAL PRINCIPLES AND STRUCTURE OF PESTICIDE MEASUREMENT SYSTEMS

Metrics based on toxicity alone vs. toxicity/exposure combinations

Risk assessment indices exhibit varying levels of complexity (Figs. 1 to 13). At the simplest level, the index for effect is based on a measure of toxicity alone. Often this is the case with point based systems, where many very simple endpoints are scored and the scores aggregated. The score given to toxicity will be incorporated in a final algorithm with scores of other variables (not shown – scoring methodologies and aggregation methods for all the risk assessment systems and indicators are detailed in Appendix A). A model such as APPLES (CCME 2004) has adopted this approach. Here, a toxicity score is combined with scores for ‘presence of active ingredient in the Canadian environment’, ‘environmental fate’, and a ‘socio-political criterion’. Other models will instead combine toxicity and exposure in a single metric. Such risk measurement indices typically become increasingly complex according to the degree of sophistication used to estimate exposure. Most European indices are simple risk ratios which resemble methods used by regulatory bodies in their evaluation process e.g. EPRIP, the Norwegian Indicator, POCER, etc... (Tables 5 and 6) More complex indices may have refined their exposure estimate by incorporating variables such as chemical half-life, crop interception, chemical solubility, drift, soil depth and density, and measures of bioaccumulation. Most complex indices require input from a separate model based on site-specific data (e.g. the earthworm index of the Dutch Environmental Yardstick system).

When and how pesticide use data are integrated into the system

A somewhat related issue is whether the cumulated quantity of each active ingredient is the starting point for the development of a risk index or whether risk is assessed on the basis of actual (from surveys) or label application rates and later aggregated by extent and frequency of treatment. In the former, sales statistics are often used as a basis for estimating pesticide use and the cumulated amount (tonnage) of the different pesticide products delivered to some component of the environment is estimated, often on the basis of a geographically defined area such as watershed, state or province, or country. Examples include the Danish Load Index (OECD

2000b, 2004) which is calculated from the ratio between the total sales and the toxicity of a pesticide – this is later summed for all pesticides. The Frequency of Application index (OECD 2000b, 2004) is also based on sales data although the final metric is not a risk estimate but only an estimate of the pesticide load.

More recently, the PEI Relative Ranking System (Dunn 2004) which was based on CHEMS (Swanson 1997) considers the volume in kg of active ingredient (approximated from sales data) but pesticide load is scored and applied as a factor to weight the toxicity of the pesticides. These systems do not incorporate application rates but handle exposure at the aggregated landscape level only. Such systems are ideal to characterize pollutant release inventories and even allow for a ranking of pesticide and non-pesticide products alike. However, as discussed by Dunn (2004) in her prioritisation of PEI pesticides, these systems do not represent true risk assessments. In order to reflect actual risk, a scheme needs to integrate toxicity and exposure at the single field application level. Also, implicit in a measurement based on sales data is the assumption that risk increases linearly with the cumulated amount of product used on a landscape. It does not consider the assimilative capacity of the receiving ecosystem and that risk may be kept low through a judicious choice of application rate. For example, systems based on cumulated pesticide quantities assume that the impact of a single application of a pesticide at 600 g a.i./ha is equivalent to split applications of 300 g a.i./ha a week apart. If we want to accurately express pesticide use as risk to the environment, it is best to adopt a system which assesses risk on the basis of a single application and integrates that risk as a last step only. Also, we believe that risk assessments based on single applications can more easily be updated to reflect product popularity in time and space, they can provide information on the relative risks of competing pest control approaches, and can provide useful information, even in the total absence of pesticide use data.

Use of exact values vs. toxicity classes

Several systems have recommended the use of toxicity classes. Generally, we have found no consensus among the choice of threshold cut-off criteria and generally no justification based on biological or ecological relevance. Often, categories were based on orders of magnitude (Tables 5 and 6). However, there is a degree of arbitrariness associated with these systems. Two products with statistically indistinguishable toxicity data points of 9 and 12 mg/kg might find themselves in two separate toxicity classes based on a system which recognizes toxicity values of 1-10 vs. 10-100. On the other hand, some systems have adopted a hybrid strategy between categorization and a continuous function. The top and low scores are fixed based on threshold values but the intermediate scores are obtained on the basis of a linear function such as a regression line (e.g. CHEMS and the related PEI model). Ipest (Van der Werf and Zimmer 1998) also uses a similar scoring method. Here, threshold values define two categories: F (favourable, no potential environmental impact) and U (unfavourable, max potential for environmental impact). When a value falls between the threshold values, Ipest uses sinusoidal-shaped functions to assign membership values between the lower and higher score, depending on the degree of membership to either subset.

We believe that the use of a scoring strategy early on in the calculations translates into a loss of information. Another limitation is that the use of toxicity classes prevents the consideration of the application rate as an important modifier of real toxic potential. We therefore favoured those indices which use exact values rather than classes, recognizing the false precision often inherent in such data points.

Choice of toxicity endpoints to drive a comparative risk assessment system

The choice of suitable input variables is a function of data availability, validity and representativeness. As noted by all authors in search of the perfect system, it is imperative that the input data be widely available in order to easily compare a large number of pesticides. This means that we favoured indices developed from data routinely available from the regulatory review process. It is important to have as few data gaps as possible in any risk ranking system so as not to compromise the accuracy of the ranking.

The toxicity endpoints need to be valid. For example, serious questions relating to the validity of the 5-day dietary test in birds have been raised (Mineau et al. 1994). These test results are subject to wide variation resulting from the latitude allowed under test conditions and the endpoint is largely invalidated by any food avoidance, a common artefact of laboratory conditions. In a regulatory context this is less of a problem because regulators have access to the raw data including the food consumption information and can therefore make allowance for this problem or choose not to use the test for this reason (European Commission 2002a). However, because only endpoints are widely reported and publicly available, a comparative risk assessment scheme could easily be misled into using invalid test results.

Also, there has been controversy surrounding no-effect data. Because of the multiplicity of possible effects in chronic toxicity tests as well as limitations related to the cost of testing, chronic test data are often not well suited to a probit (ECx) approach and no-effect levels are usually reported and used for regulatory purposes despite abundant criticism (e.g. Chapman et al. 1996; Crane and Newman 2000). Because no-effect data are, and will continue to be, available in the near future, long-term risk assessments have been (Tables 5 and 6) and will likely continue to be based on them. The choice of input variable in consideration to specific taxa is discussed in section 3.

Manipulation of toxicity data.

Interspecies variation in toxicity is an important source of variation in any pesticide risk assessment measure. Even for a single test species some inter-test variation is the norm e.g. variation in test results from many laboratories, or simply due to differences in results from many replicates. Four main approaches have evolved to deal with cases where many data points are available:

1. Restricting data input to a single species or to a few defined species

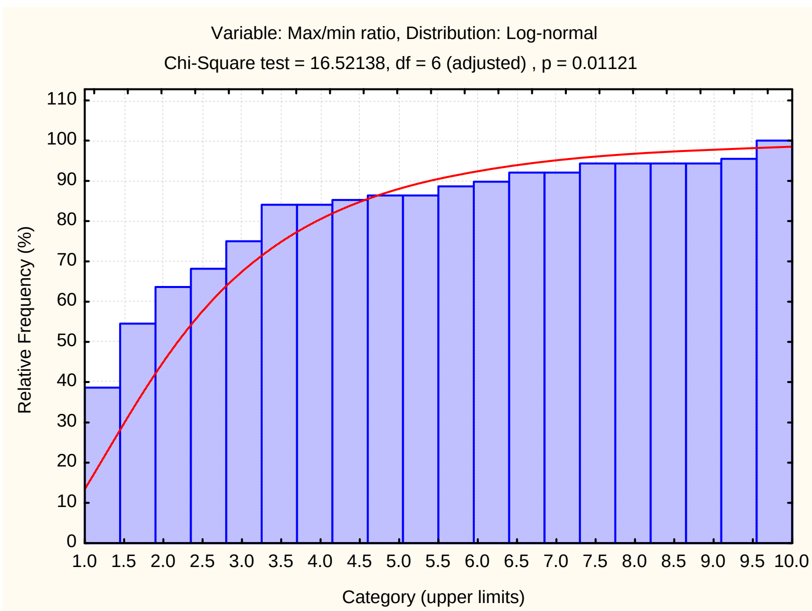
- This approach is reasonable when data are unlikely to be available for a large number of individual species. For instance, the honeybee is the only pollinator to be routinely tested although data from other pollinator species may be available on a case by case basis. In this particular instance, the main question to ask is to what extent the honeybee is representative of other invertebrate species, starting with other pollinators and then extending more broadly to other invertebrate taxa. If bee test data is being used as a measure of wild pollinator impact or of impact to terrestrial arthropods at large, then certain strategies adopted for some indices such as accepting a default value of zero for risk to bees in certain conditions e.g. if crop is not at a flowering stage (Table 5c) are not reasonable. In the case of mammals the logical choices are limited to rats and mice – more values being available for the former. Strain and/or sex differences are ignored although this can be an appreciable source of variation. For birds, the two standard test species are the Northern Bobwhite and Mallard although many more species have been tested, especially in the case of older products. The ATRI index incorporates acute toxicity data for partridge only, as it is known to forage in arable areas and because pesticides are linked to its decline (note that in the ATRI system, there is no index for mammals because authors consider that birds are generally more sensitive than mammals). Red and Grey Partridge data were commonly generated by the French department of agriculture but we do not believe this program is currently active.

- On the other hand, regulatory guidance (e.g. European Commission 2002b) cautions against over reliance on a single indicator species, especially in the case of new chemistry pesticides with very targeted modes of action (see discussion of this point in section 3.2.b).

2. Taking the lowest value from all available data for a given taxon.

- Whether in the U.S. or the EU, it is common for regulatory assessments to be based on the lowest available toxicity value and this practice has made its way into several assessment systems, for example, p-EMA, Ipest, the Dutch Yardstick, POCER and APPLES (Tables 5 and 6). Unfortunately, this approach introduces a major element of stochasticity and unfairness into the system. The more species are tested with a given pesticide, the higher the chances that a sensitive species will be found. In comparing older and newer pesticides, this immediately places older products (for which test data have accumulated over the years) at a disadvantage. The APPLES system currently used by Environment Canada to prioritize pesticides suffers from this problem. The use of the 'lowest available value' also is a strong disincentive to industry to test any more species than they absolutely have to; or to withhold supplementary data.

- To see how much uncertainty could be introduced by always taking the lowest available data point where more than one test was available for a single test species, we looked at 88 pairs of avian LD50 data. Pairs were chosen on the basis that the endpoints all came from full probit–type studies (as opposed to up and down approximate LD50 values) and all were scrutinized in order to ensure they represented truly different tests rather than re-analyses of the same data with different probit models, a common artefact of toxicity databases. The cumulative distribution of the maximum over minimum values is plotted here:



- Just a little over 50% of all paired comparisons were within a factor of 2. Only ¾ of all paired comparisons were within a factor of 3. Clearly, uncertainty of this magnitude could play havoc with a toxicity-based risk assessment. To always take the lowest available value would exacerbate the ad hoc nature of any comparison of alternative products. Where 'the lowest available value' rule also applies to different species, then the potential biases become all the more serious.

3. Using a distribution approach to derive a defined toxicity data point.

   - The simplest distribution-based toxicity value is a measure of central tendency. Thus, certain indices make use of mean toxicity values although medians may be more appropriate. Unfortunately, whatever information is available on interspecies variance is lost in the derivation of a mean or median. Hence the development of a species sensitivity distribution (SSD) from which any value – usually a defined tail of the distribution at the sensitive end – can be derived. The topic of SSDs has been amply reviewed and a detailed discussion is beyond our requirements. However, it is important to recognise that different approaches are more suitable for different sizes of datasets. Large and diverse datasets can have their values ranked and plotted in order to define a distribution tail e.g. the 5% tail is often chosen. The values can be bootstrapped in order to 'generate' a larger dataset without presupposing an underlying distribution. This method does tend to reinforce biases present in the initial dataset but it does allow for interesting weighting of toxicity values (see approach by Duboudin 2004 – reviewed below). Other methods (Aldenberg and Slob 1993) are more suitable for smaller datasets and introduce sample-size based extrapolation factors which can be applied to a distribution variance after a theoretical distribution has been hypothesised (e.g. log normal or logistic). Finally, new methods based on Bayesian principles are being developed. The use of SSDs is reasonably well developed for aquatic taxa as well as for birds and mammals. The various methods proposed to date for birds and mammals have been reviewed recently by Luttik et al (in press).

4. Weighting of toxicity values

- In the usual SSD, each data point (usually toxicity values for different species) contributes equally to the resulting toxicity benchmark – whether geometric mean, median or defined tail of the distribution. It is possible to give a different weighting to each available data point in order to derive a more 'meaningful' benchmark. The EcoRR system (Sanchez-Bayo et al. 2002) suggests weighting the geometric mean by a biodiversity factor (Table 5a). The geometric mean of toxicity values for a given taxon will be weighted by the number of species of that taxon relative to the total number of species present at an ecosystem level. This is equivalent to the 'enviro approach' described by Duboudin and colleagues (2004) for aquatic data. Duboudin et al. (op. cit) recognised two other approaches – the 'biblio' approach which takes all available data as it comes and therefore is essentially unweighted, as well as the 'equi approach' which gives equal weight to each taxonomic group regardless of the number of values available for each. We would like to propose another possibility for weighting of toxicity data – weighting by the ability of the species to affect a population recovery following a pesticide impact (see 3.2.f for a fuller discussion).

Use rates and formulation specific issues

The same pesticides are often used in a variety of crops, at different rates and in different formulations. Following the age-old principle that the 'dose makes the poison' (Paracelsus) we favour those systems which included as much information as possible on application rates and other product specific information. It would be difficult to argue that the clay-based granular formulation of an insecticide has the same potential impact as a foliar spray of the same a.i. to pollinators. Yet, the vast majority of indices do not explicitly deal with formulation issues. For most systems, toxicity is calculated only for technical products. The model we are proposing for birds (Appendix B) calculates the probability of bird mortality based on field data from liquid formulations but then applies a series of weighting factors which account for differences in formulation. This is the approach currently being developed by Benbrook, Curtis and others under the PEAS system (see introduction to Appendix B). Others on the other hand (e.g. Dunn 2004) have included both results of tests using formulated products and active ingredients in the geometric mean of toxicity values. Prior to calculations, Dunn (op. cit.) has adjusted formulated products to reflect the percent concentration of active ingredient in the product. Unfortunately, there is no way to adjust for formulation effects on absorption, pharmacokinetics, etc. She found that when a toxicity value was available for the active ingredient, it generally was either greater than or within the same magnitude of the geometric mean determined for the pesticide using both formulated and active ingredient studies. She also found that using the geometric mean over toxicity values derived from the active ingredient was more conservative in some cases. This approach needs further consideration and consultation.

Handling of missing data

Missing data present a main obstacle to the development and use of any system. The types of scoring criteria used and the design of the final algorithm may depend on whether experimental data are available or can be estimated with an acceptable degree of accuracy. A few strategies were identified to deal with missing data, although such information was not often reported. These are:

1. Replace missing data by default values

   - This is likely the simplest approach. Missing scores can be replaced by a default score value so that no given variable will be excluded from the final algorithm. For instance, default scores can be those of other species (e.g. the score for bees filled gaps for missing arthropod data in PRIHS-1). Data gaps can also be filled with the mean scores for similar class of chemicals, although some have argued against this approach (Levitan 2000). Alternatively, default scores can be at the lowest end of score categories (e.g. use score of 0 if products are presumed to have no negative impact) or at the high end of a score category (e.g. highest score given to herbicides when assessing effect on plants). Some have chosen to fill gaps with the median score (e.g. the University of California index).

- Swanson et al. (1997) examined the validity of the use of a default score. Their algorithm was run both with default hazard values of zero and five (the minimum and maximum scores) for each missing data point of 3 variables including fish toxicity. The final ranking order changed for 6 of the 30 tested chemicals (note that these are not all pesticides) and the top 11 chemicals kept the same order regardless. Their system used a combination of toxicity classes and continuous values – it isn't clear whether this would hold for other types of systems.

2. Tiered approach

- A tiered approach implies that certain endpoints are used preferentially and if these are not available, the range of acceptable data will widen. In APPLES, the preferred acute data for fish, aquatic invertebrates, and algae were respectively: the 96-hour rainbow trout LC_{50}, the 48-hour Daphnia magna EC_{50}, and a green algae (Selenastrum capricornum) EC_{50}. If these were not available, they would accept results from a test performed with the same exposure period but for another similar species (e.g. a 96-hour LC_{50} for another salmonid). Ultimately, results for tests for another species but with any exposure period could be accepted. The CHEMS model (Swanson et al. 1997) accounts for data gaps with a similar approach. For the terrestrial compartment, a tiered approach is proposed for p-EMA.

- Filling data gaps by widening the pool of available input data may not be ideal. Inter-species differences have been shown to be significant, even between related species.

3. Estimate missing data using a QSAR

- A QSAR is a predictive relationship between descriptor measures (often the physical properties of compounds) and the response of the biological system under consideration. It is reported that acute fish LC_{50} can be accurately estimated by using a QSAR as a function of the octanol-water partitioning coefficient (K_{ow}). More realistically, QSAR approaches can be used to help generate correction factors that could be applied to sub-optimal data e.g. static vs. flow through tests, 48h toxicity to 96h toxicity etc. For example, equations have been proposed to estimate missing fish NOEC using a QSAR based on K_{ow} and the 96-hour fish LC_{50} (Swanson et al. 1997):

  i The NOEL values for organic chemicals were calculated using a continuous, linear function: $NOEL = LC_{50} / (5.3 * \log K_{ow} - 6.6)$ for $2 \leq \log K_{ow} < 5$

  ii Organic chemicals with a log K_{ow} greater than or equal to 5 are generally more toxic to fish and were assigned a lower NOEL value: $NOEL = 0.05(LC_{50})$

  iii Organic chemicals with a log Kow less than or equal to 2 are poorly fat soluble and were assigned a higher NOEL value: $NOEL = 0.25(LC_{50})$

iv Because inorganic chemicals are poorly fat soluble and their fish toxicity does not correlate to log K_{ow}, the NOEL values of the inorganic chemicals were based entirely on the fish LC_{50} values.

On the other hand, the OECD Aquatic Expert Group (OECD 2000) discusses two ways to extrapolate from acute to long-term toxicity data to fill data gaps when long-term toxicity data is lacking: (i) by dividing by standard factors, such as those used in a registration procedure, and (ii) by using the mean of the ratios between acute and long-term toxicity data for the same taxon from other substances of the same class or group of chemicals. According the Expert Group, as long the same method is consistently applied, the particular method chosen would not affect the indicator's ability to rank products. This is true, but our ability to obtain a ranking is not the ultimate goal; rather, our rankings need to reflect actual pesticide risk if the intent is to direct and/or modify the behavious of pesticide users as a result of those rankings.

More work is clearly needed on the issue of missing data. As part of the NAESI project, a large database of toxicity data is being assembled in order to perform some exploratory analyses and propose empirically-based strategies for missing data.

True risk vs. hazard as defined by a toxicity-exposure metric (validation)

In our opinion, this is probably the most critical element of any risk assessment scheme, yet it is consistently ignored. With several risk measurement systems, risk is assumed to vary in direct proportion to a linear combination of toxicity and exposure. However, it is rather improbable that true risk is linearly related to a toxicity-exposure metric regardless of how the latter is calculated. For example, an insecticide with an avian LD_{50} of 0.6 mg/kg may not be twice as safe to birds as another with an LD_{50} of 0.3 mg/kg (assuming equal application rates). This is because either may be sufficiently lethal to the majority of exposed birds that the slight improvement in toxicity is inconsequential. In the same way, a compound with an LD_{50} of 600 mg/kg may not be any safer than one with an LD_{50} of 300 mg/kg because neither compound poses any risk of mortality. On the other hand, there is an expected range of toxicity over which we expect risk to vary in a linear fashion – somewhere between the two extremes described above

To avoid this problem, the final exposure-toxicity metric can be categorised into risk classes (as opposed to toxicity classes as discussed in section 2.6). This is often done in regulatory assessments where significance is attached to whether a toxicity/exposure ratio (also known as a risk quotient) falls above or below some regulated cut-off, such as the number 10 proposed by the Uniform Principles of the European Council Directive 91/44/EC (European Union 1997). This cut-off is usually a nice round number but, unfortunately, has no basis in science. Indeed, most of the usual toxicity-exposure metrics have not been validated against any real world predictions.

One notable exception among all the risk assessment systems that were reviewed is the proposed measure of avian acute risk (Mineau 2002). This index takes the usual toxicity-exposure metric but translates that into actual risk (the probability that a field will sustain avian mortality) on the basis of available field studies. Prediction ability is enhanced by finding other predictor variables such as combinations of physico-chemical constants reflecting dermal penetration ability, etc. Ideally, we would like to find such a field validated approach for all environmental components. Still, it is important to note that, although the avian index of risk of mortality has been field-validated, it is only one aspect of the total risk of pesticides to birds. We are still unable to validate either a chronic or reproductive impact, a sub-lethal acute effect leading to long-term debilitation or mortality or even less an indirect impact.

We are unlikely to find a large number of field studies for most of the environmental components of interest. In the absence of a large pool of field studies, there are other approaches that can be pursued. The first is a consideration of incident data. These can be used to set benchmarks for risk indices. Such an approach was used for bee kill incidents in the UK (see section 3.1.c) and could be extended to other sectors – e.g. fish kills. Another possibility is to use the few available field studies to set benchmarks. This approach was followed by Mineau and Duffe (2001) for birds before the models described in Mineau (2002) were developed. Risk indices associated with specific incidents were used to infer lethal risk with other specific untested pesticide uses. Similarly, Sheehan et al (1995) developed benchmarks of acceptability for pesticide impacts on prairie slough based on the loss of invertebrate biomass that would be sufficient to affect consumers. These approaches may not provide for a nice linear scale of pesticide indices but they do allow picking an empirically determined level of acceptability rather than an arbitrarily-chosen value of 10 or 100.

Much of the work we envision in subsequent years of this NAESI project will deal with such validation attempts. We are in the process of putting together datasets that will allow for such attempts at validation. Again, if we hope that these risk indices will be used to set standards of best agricultural practice, we need to be able to demonstrate that reductions in the risk scores will lead to meaningful environmental improvement.

Combining and weighting individual risk indices

Risk assessment systems use different levels of aggregation in order to characterize risk to a component of the environment e.g. the terrestrial environment. It is valid to ask whether we should consider both the risk to birds and to wild mammals when one could suffice. Developing too many indices will make product comparisons potentially more difficult and confusing. One reason for including both is if they are differentially affected and vary in their susceptibility to different pesticides. It is known for instance that birds have a very high susceptibility to organophosphorus pesticides because of their inability to deal with the toxic oxon metabolites. A case in point is diazinon, an organophosphorus insecticide which, until recently, was considered safe enough to be allowed on to the domestic market while at the same time having one of the worst environmental record with respect to bird mortality. On the other hand, mammals tend to be more susceptible to the neurological effects of pyrethroid insecticides than birds. This argues strongly in favour of keeping both taxa when trying to quantify the risks to the terrestrial environment. Similar arguments can be made for any pair of taxa we care to consider.

We then have the choice of presenting the risk to birds and mammals separately or of including both in a single index. For example, PERI will give each component equal weight by calculating an average of the scores that were obtained for all terrestrial taxa. In contrast, the Norwegian Indicator will consider only the highest risk component in the score for terrestrial taxa. Others will use different weighting factors which are multiplied to the score as these are combined. The EIQ index gives more weight to beneficial arthropods than to birds and bees, an understandable choice given that it was developed to promote IPM strategies. The disadvantage of this approach is that the end result is highly dependant on the weighting which itself is a value judgment, forcing one to weight one environmental component against another. Also, without the exact knowledge of the component which is responsible for a bad score, it is difficult to propose mitigation. A high pollinator score is mitigated differently than a high terrestrial invertebrate or wild mammal score. We believe that in the spirit of APF, it is critical to keep the separate risk indices separate and not to combine them. As to how many different environmental components need to be assessed, that decision is likely to be made on the basis of data availability rather than on more scientifically defensible grounds. Some critical components of the ecosystem are missing completely for all of the indices examined, e.g. amphibians and reptiles. It is important to assess to what extent these taxa may be subsumed under other existing indices.

CHOOSING THE BEST OPTION FOR EACH ENVIRONMENTAL COMPARTMENT

Terrestrial biota

a. Mammals

Toxicity measure

Based on the discussion above, certain indices are found to be more suitable than others. We do not favour indices which use toxicity classes (section 2.6) such as PESTDECIDE and have argued against using the lowest toxicity value, a strategy favoured by the p-EMA system (Table 5a).

A distributional measure, such as the HD_5, would be the best option in order to properly address inter-species differences in susceptibility (section 2.4). However none of the models we have reviewed for mammals has proposed the use of such a measure. Luttik and Aldenberg (1997) have proposed factors that could be applied to small numbers of toxicity values for mammals. The method entails applying an extrapolation factor to the geometric mean of the available toxicity endpoints. Unlike the situation in birds, acute toxicity values need not be scaled for bodyweight (Sample and Arenal 1999; Luttik et al. in press) to account for inter-species differences in mammals. Sample and Arenal (1999) have calculated an average scaling factor of 0.94 for mammals. A scaling factor close to one implies that an adjustment based on the way toxicity scales to body weight would not be required for mammals.

Some have proposed risk quotients based on the dermal LD_{50} rather than the oral value (PRIHS-1 and EcoRR for the soil compartment). These address exposure from contaminated soil only (fossorial mammals?), even though dermal exposure can occur through other routes (i.e. contact with vegetation). Mineau (2002) has demonstrated that dermal exposure was important in determining field impacts of toxic insecticides in birds. There is no reason to believe that the situation is not the same in mammals. Also, dermal toxicity data are generally available for mammals (rat, rabbit and occasionally mouse) unlike the situation in birds. Indeed, based on physiology and habits, dermal toxicity is likely more important in mammals than in birds. However, we lack the pool of field studies to validate a dermal index. As with birds, the potential relative importance of dermal toxicity could be assessed using a simple ratio of dermal to oral LD_{50} (the dermal toxicity index or DTI).

Exposure measure

Proposed systems to date rely on a RUD (Residue per Unit Dose) measure as defined in regulatory evaluation guidance (e.g. European Commission 2002). In collaboration with the PMRA and in light of the most recent re-analyses of plant residue information (Baril et al. in press), we will propose an exposure scenario that best reflects available science. We will also explore the availability of dermal exposure models (from mammalian regulatory evaluations) in order to obtain estimates of both oral and dermal uptake. With the full industry data package, and a full probit model (as opposed to an approximate LD_{50}) each of these can be translated into a probability of mortality. A method would then be needed to combine the results of the oral and dermal probabilities e.g. oral dose x equates an oral LD_5 and dermal dose y equates a dermal

LD_{15} – what is the combined probability of mortality. Alternatively, should probit slopes not be available, two toxicity/exposure ratios (TER or HQ – Hazard quotient), for both oral and dermal toxicity could be generated for scoring.

Application factors as developed for bird will need to be developed here (section 2.6). For instance, it is doubtful that mammals are attracted to granular pesticide formulations as are birds. On the other hand, as discussed by Hart et al.(2003) with respect to p-EMA, granules may be ingested accidentally when adhering to prey items – e.g. earthworms.

Validation potential

Even if few mesocosm or full field studies exist for this taxon, the few existing field studies could possibly allow for the development of field-bases thresholds of acceptable vs. non acceptable risk for a given product (section 2.8) with or without consideration of dermal exposure. The strategy of establishing benchmark pesticides is one which could be adopted where field data do not allow for full validation.

Database needs and progress to date

A mammalian acute toxicity database has been assembled. Analyses of the database to begin summer 2005.

b. Birds

The risk assessment procedure for birds is presented here as an example of the only validated risk assessment tool to date, i.e. where we would like to take all other environmental components.(See Appendix B)

c. Bees

Toxicity and exposure measures

Most indices for bees are very simple hazard quotients, based on LD_{50} (Table 5c) either oral or contact where the exposure compartment is based solely on the application rate of the sprayed pesticide. The Italian Bee Risk Indicator (BRI) is distinctive because it was designed to estimate risk to bees from exposure to contaminated pollen rather than direct exposure from spray. Equations are proposed to calculate the amount of pollen ingested as well as the amount of pollen picked up by bees to address both dermal and oral toxicity but only from contaminated pollen. It might be possible to integrate this model with the more standard ones based on droplet toxicity although the pollen model is quite complex. It may be very important to consider pollen ingestion in the case of micro-encapsulated pesticides given the history of bee kills with micro-encapsulated organophosphorus products.

Validation potential

We consider hazard quotients to be a good starting point for the NAESI project. These are successfully used for risk assessments in Europe and the threshold values which lead to categorisation of a pesticide according to level of concern were validated with bee incident records (Aldridge et al. 1993). However, based on a pers. comm. from one of the authors of this validation paper (A. Hart), we believe the analysis has to be repeated in order to confirm hazard ratio trigger values. Two complicating factors have been uncovered with respect to the estimation of bee (pollinator at large?) risk. The first is the synergism between different pesticides e.g. conazole fungicides with OPs – we may not be able to address this issue in a scoring system; the second is the possibility that the overall risk of pesticides to bees may have a non-monotonic relationship to toxicity. The reason for this is that a highly toxic pesticide may kill foraging bees on the spot while less toxic products or those with a delayed mode of toxic action allow the foraging individuals to return to the hive with contaminated pollen or nectar and, potentially, kill the brood – a much more serious outcome. More consultation and development are required here.

Database needs and progress to date

We have obtained a recent download of bee incidents and will be using this to validate the indices. To refine terrestrial invertebrate risk assessments, the relationship between bee morality and that of other beneficial arthropods should also be investigated (section 2.4). Because bees remain the largest source of data, it is important to know how broadly they represent terrestrial invertebrates.

d. Beneficial arthropods

Toxicity and exposure measures

Many types of variables were considered by different indices (Table 5d): 'toxic impact on beneficial arthropods', 'reduction in the control capacity of beneficial arthropod', 'arthropod inhibition', etc. In most cases, variables are scored, often based on a 30 % effect threshold value as was historically recommended by previous risk assessment guidelines. The use of such a threshold has been criticized for risk assessments (Campbell et al. 2000) as it would generate too many false positives, leading to excessive unnecessary higher tier testing. In addition, the relevance of this trigger value to protect arthropod populations has not been validated.

The most promising system is the European ESCORT 2 index. Here calculations are based on the LR_{50} (lethal rate causing 50% mortality), which is derived from a standard dose-response laboratory test. Because this model is based on a hazard quotient approach, the need to use the arbitrary 30% effect threshold value is eliminated. Calculations are performed using data for two indicator species Aphidus rhopalosiphi and Typhlodromus pyri. These test species have been shown to be suitably sensitive, at least relative to other common test species (Candolfi et al. 1999). Based on field work done by Campbell et al. (2000), trigger values of ≥ 12 for T. Pyri and ≥ 8 for Aphidus spp are proposed.

Our main problem in Canada (and N. America more generally) will be the lack of data for beneficial arthropods. European jurisdictions have been requiring these for a while but they are not widely available here.

Database needs and progress to date

Data availability is being explored through U.S. and open EU sources. In addition, we are attempting to obtain a full version of the very large SELECTV database from Oregon State University in order to carry out our own analyses. As a first step, we will carry out comparisons of field outcomes with honey bee toxicity data.

e. Earthworms

Toxicity and exposure measures

Indices based on toxicity classes such as PERI (Table 5e) have not been retained (section 2.6). The other surveyed indices are risk quotients based on the LC_{50} (Table 5e).

The p-EMA system favours data for Eisenia foetida, even though we also found data for Lumbricus terrestris. An interesting observation was made when reviewing field studies (Van

Gestel 1992). It was found that the field toxicity was also related to earthworm behaviour. Individuals of the Lumbricus terrestris species were found to be more susceptible to pesticide exposure because these feed near the surface of the soil, where the concentration of pesticide is likely to be higher. A simple geometric average is probably appropriate if LC_{50} values are available for a few species only.

The toxicity to earthworms has been shown to be affected by the soil properties. Lipophilic organic chemicals in particular will bind to soil organic carbon and be less bioavailable. The artificial substrate of the earthworm laboratory test has higher organic matter content than many natural soils and as a result, the LC_{50} may be higher than it would be in natural soils thereby giving the impression that the test chemical is less toxic. To account for such a difference in risk assessments, the European Commission (2002) suggests dividing the LC_{50} by 2 when the $\log K_{oc}$ is >2.

The available models distinguish themselves through the complexity with which soil/pesticide application dynamics are modeled. Some (e.g. the Norwegian model, SYNOPS) emphasize application variables such as crop interception; others (e.g. the Dutch yardstick) emphasize soil properties and soil/pesticide interactions through physico-chemical measurements such as Koc, soil DT_{50} etc…

<u>Database needs and progress to date</u>

Ideally, risk should be assessed on the basis of crop conditions specific to the Canadian environment. The PMRA is currently building a series of crop scenarios broadly representative of Canadian cropping conditions. As with several of the aquatic assessment systems (see below), we believe it would be most efficient to use the 'official' crop scenarios to produce a series of input values to use in the appropriate soil persistence and distribution models. In this case, the better developed models to emulate might be the Dutch Yardstick model with its attendant soil model. Unfortunately, validation data are not likely available to assess how many soil variables need be considered in order to properly estimate risk. In the case of the earthworm, however, field conditions can so easily be replicated in the laboratory that validation may be suitably performed on laboratory data. A full literature search is needed. Finally, we need to consider whether all earthworm data points are comparable. Test conditions have changed over the years and this may make comparisons difficult.

f. Terrestrial plants

The only index we have surveyed for terrestrial plants is highly insufficient (Table 5g). It is based only on a coarse indication of a chemical's phytotoxicity using two categories: phytotoxic and non phytotoxic. Also, details on the nature of the data (e.g. which plant species, which endpoint) and the cut-off threshold value are not cited. An index will therefore need to be developed.

In Canada and the United States, phytotoxicity data is provided to regulators for risk assessments. Under the OECD and EPA guidelines, testing for regulatory purposes is performed on crop species. Current guidelines recommend that testing be performed on at least 6 species from different taxa. Perhaps this will allow for a distribution-based method for plants. However, whether or not the use of crop species in risk assessments is in fact representative of the plants to be protected has been recently discussed (Boutin and Rogers 2000; Boutin et al. 2004). It was shown that crop species are often less sensitive than wild non-target species and that risk assessments based on results under the current guidelines are not adequate for the protection of off-field plants. Therefore the selection of species currently favoured by regulatory bodies (i.e.

crop species) will very likely cause an unacceptable bias toward underestimation of risk (Boutin et al. 2004). Because few data on off-field species are currently available we can only recognize this limitation and try to compensate through the development of extrapolation factors. Current research underway at NWRC (C. Boutin pers. comm.) may help in this regard.

Formulation is also an issue when dealing with toxic effects in non-target plant species. Even more than in other organisms, the dose of herbicide to which non-target plants are exposed to is not necessarily predictive of the intensity of the effect because the actual herbicide uptake depends on many factors related to plant morphology, environmental factors, factors related to product application, etc. (discussed in Boutin et al. 2004). The use of surfactants in a herbicide formulation often increases herbicidal activity by improving spreading and retention of the herbicide on the leaf surface, increasing the uptake or translocation within plants. It could therefore be argued that our index should include formulation data since an index based on data for the active ingredient will likely underestimates actual phytotoxicity. This will depend on data availability.

In fact, the need for formulated data runs through several (all?) of the other environmental components. Unfortunately, this is a severe data gap in most cases. Also, it is often impossible to know which specific formulation will be used, how different formulations will be tank mixed and whether adjuvants will be added to existing formulations. This is a long standing problem in risk assessment which will not be solved through this exercise.

g. Long-term indices for terrestrial components

Two types of long-term indices were surveyed. The PRIES-2 model has proposed to score the NOEL for mammals, birds, bees, and beneficial arthropods. For earthworms, the NOEC is scored. The limitations associated to the use of toxicity classes was discussed in section 2.4. The limitation of the NOEL (as opposed to an ECx value) has also been discussed. Another model (the PRIHS-2 model for mammals) is based on a toxicity/exposure measure. Limitations of the PRIHS-2 model are related to the measure of exposure (dermal exposure) which incorporates site-specific data (e.g. soil depth and soil density). We recognize that default values could be used if site-specific data are not available.

Chronic NOECs are generally available for both birds and mammals. However, their interpretation is problematic and has been the subject of an extensive review by the UK government (Bennett et al. in press; Mineau in press). Following an expert group meeting in 2004, new evaluation methods were proposed to handle the results of current reproduction tests in birds and mammals. More work is needed to see whether and how this could be integrated into this exercise.

Aquatic biota

<u>Toxicity measure</u>

a. Fish

Such systems as the PEI model and APPLES have considered data from a single species i.e. rainbow trout (Table 6a). Because rainbow trout data is required for product registration both in Europe and the United States, it can be argued that the choice of this species is driven by widely available data. Massive datasets exist for other fish species (e.g. fathead minnow) but data may be lacking for newer pesticides. Data for other fish species are available on a case by case basis. For product registration, tests on a warm water fish are also required (specifically the bluegill for the U.S. – which is consistent with the choice of species for the MATF model). The list of commonly tested species also includes other salmonids and species such the goldfish, the common carp, the white sucker, the channel catfish, etc. (Coney 1995). We therefore believe that an SSD approach which accounts for interspecies sensitivities is feasible for fish. Most of the surveyed models have recommended the use of the 96-hour LC_{50} endpoint for fish.

b. Aquatic invertebrates

Similarly to fish, the surveyed indices have generally restricted data to Daphnia only (Table 6b). Data for Daphnia are also required for product registration and are thus widely available. But also, it has been demonstrated that Daphnia was suitably protective for aquatic insects and other invertebrates for most insecticides (Brock et al. 2000 a and b). For organic chemicals including a range of pesticides, Daphnia magna is usually among the most sensitive species (Wogram & Liess, 2001). Even when there are more sensitive groups (certain classes of insects), these are generally less than an order of magnitude more sensitive than Daphnia. However, there was a word of caution in the recent EU guidance on aquatic risk assessment (EC 2002a). Some new pesticide chemistries with very specific receptor-mediated modes of action have given very divergent results in different invertebrate classes. This guidance document recommends further testing if a product is found to have a suspiciously low toxicity to Daphnia (e.g. > 1 mg/L for a 48 h LC_{50}). Based on recent experience with neo-nicotinoids and insect growth regulators, Chironomus riparius has been recommended as an alternate species to Daphnia. Such case by case consideration of risk is not possible in a comparative risk scoring scheme. Again, a better solution may be the generation of an HC_5, any excessive variance between two study points resulting in a lower predicted HC_5.

Typically, test duration is 48 hours for invertebrates. This allows daphnids to go through one molt, at which time they are usually most sensitive to toxicants, but does not prolong the test to the point that starvation becomes a major factor (Cooney 1995). We did not find consensus on the choice of endpoint however. Some models have proposed the LC_{50} but the majority have proposed the EC_{50} as death and immobilization are most often tested. For some species of invertebrates, death is not easily distinguished from immobilization and the EC_{50} is often determined rather than an LC_{50}. For daphnids and midge larvae, the EC_{50} is usually based on death plus immobilisation. Some also believe that such a defined EC_{50} better reflects the total severe acute adverse impact than does the LC_{50} (Stephan et al. 1985). The PEI model has incorporated this endpoint. In practical terms, the use of this variable is not yet clear (i.e. are LC_{50} data and EC_{50} values considered equivalent in the calculations?)

c. Algae

The p-EMA model has proposed the use of the lowest species (section 2.4) The APPLES model preferred tests performed on Scenedesmus, or Chorella. The latter is in agreement with OECD recommendations. Brock et al. (2004) have found that, based on their position in the SSD of primary producers, the OECD recommended algae (i.e. Scenedesmus subspicatus, Selenastrum capricornum, Chorella vulgaris) are in general suitable for estimating risks to aquatic plants, especially when a safety factor of 10 is applied. On the other hand, Lewis (1995) has given ample examples of inter-species variation for algae and an SSD approach may be possible.

There is still some debate about which endpoint should be used in some tests. For example, for algal tests, the European Commission (2002a) suggests using the lowest of either biomass or growth endpoints for risk assessments. There is an ongoing debate about whether growth rate (which might be ecologically more relevant) or the more sensitive of the two endpoints should be chosen (OECD 2000). This leads to a discussion of which endpoint is the most representative of the actual impact on the exposed community at large.

d. Aquatic macrophytes

A serious limitation with this group is related to data availability. Only relatively recently have tests on macrophytes been a requirement for risk assessments (usually Lemna sp.) and toxicity data are not readily available. Even though we recognise that aquatic plants are an important part of the aquatic ecosystem, we may not be able to implement an index due to limited data.

e. Long-term assessments for aquatic components

The models we have surveyed were either based on the NOEC or NOEL. For fish and invertebrates, these were often rather complex indices because of the required use of a second model to estimate exposure. The U.S. water quality criteria are currently derived from the maximum acceptable toxicant concentrations (MATC) as the geometric mean of the NOEC and LOEC from the most sensitive endpoint (Stephan et al. 1985).

For fish and aquatic invertebrates, results from full life-cycle tests that expose all critical life stages (i.e. development, growth, and reproduction) are recommended. These are more appropriate than results from short-term chronic tests which are sometimes performed for economical and practical reasons (e.g. a 4 to7-day fish test which include only one life-stage). Short-term chronic test results do not accurately predict chronic toxicity (Cooney 1995). A short-term chronic toxicity test result is correlated to chronic toxic effects but the error in correlation may be large. For chronic exposure, flow-through tests are preferred because of the relative stability of test conditions, particularly for some highly volatile, hydrolysable, or degradable materials (Stephan et al. 1985).

f. Test conditions in aquatic testing

Beyond differences in the choice of the test species and the type of measurement are differences in test conditions which can also influence the outcome of a toxicity test (Cooney 1995). Unfortunately, test conditions were not discussed by index authors. For instance, whether results from static or flow-through tests were preferentially used for aquatic toxicity index calculations was never reported, even though it is known that problems may arise from testing performed under static conditions, especially with hydrolytically unstable, volatile or hydrophobic molecules. Also, the test material could have a high biochemical oxygen demand (BOD) so that the toxicity is masked by depletion of dissolved oxygen from the test solution. The stress caused by low dissolved oxygen could give the impression that the test chemical is more toxic than it actually is. Also under static conditions, excretory products from the test organism accumulate and may react with the test material. Adding to the potential diversity of test results are renewal tests where a proportion of the water is replaced on schedule.

Many other factors may affect the outcome of toxicity tests. Biotic factors such as life stage, size, age, disease, and nutritional status of the test organism may affect results. The same is true in regards to abiotic factors such hardness, alkalinity, specific conductivity, pH, temperature, dissolved oxygen, etc…. Control of pH in the course of testing is a very significant issue with respect to the dissociation constant of the pesticide. However, despite their importance, these confounding issues tend to be ignored because results are very rarely reported with such detail on test conditions. Typically, this is less of a problem for regulatory tests performed according to fairly strict protocols. For this reason, data vetted by regulatory bodies (e.g. USEPA, EU member country etc.) should be used preferentially, rather than test data reported in open data sources such as the Pesticide Manual (Tomlin 2000) or MSDS data sheets. However, any SSD approach will require that these more uncertain datapoints be used.

Exposure measures

Again, the main differences between competing risk assessment models are in the way aquatic exposure is calculated – from the simplest Dutch Yardstick which only uses application rate and method to more complex modeling exercises. As mentioned above (see earthworm section 3.1.e) it would be advantageous to coordinate exposure scenarios with those being developed by the PMRA for regulatory assessment purposes. The PMRA is currently building a series of crop scenarios broadly representative of Canadian cropping conditions and applying standard drift and runoff models in order to generate exposure scenarios.

Bioaccumulation

At the Copenhagen workshop on indicators (OECD 1997), it was recommended using bioaccumulation factors (BCF) or the log K_{ow} as a measure of potential bioaccumulation. However, inclusion of a BCF in the exposure-toxicity ratio approach would complicate the algorithm considerably (OECD 2000). The scoring approaches could, however, more easily accommodate the log K_{ow} which is both a measure of bioaccumulation and a factor in the context of exposure and mobility (discussion on if/how to separate those e.g. by including the K_{oc} is still ongoing). Although not shown in tables, some have included BCF in their model (e.g. the PEI model; refer to Appendix A).

Incorporating recovery in risk indices

At an ecosystem level, it has been proposed that the impact of disturbances caused by non-persistent pesticides depend not on the short-term effects of exposure on sensitive organisms, but on the intensity and frequency of the exposures relative to the rates of recovery of the populations (Barnthouse 2004). The latter suggested that risk assessments could be improved by incorporating population recovery rates. Barnthouse therefore developed a model to estimate recovery times from life history data. He found that the generation time was by far the most influential variable. Generation time is defined as the average time interval from the birth of an organism to the birth of its offspring. Recovery times were found to be greater for species with a long-life cycle. The growth rate of a population was not found to be as significant as the generation time. Thus as a rule of thumb, populations of animals with large body sizes should recover much more slowly than those with small body sizes. Also, by comparing population recovery rates calculated by using the generation time-based model to those observed in mesocosm studies for invertebrates, he found that actual recovery times were generally shorter and more variable than model predictions.

The possibility of incorporating recovery in our models should be further investigated. One possible approach is an extension of the SSD strategies presented by Duboudin et al. (2004). As described earlier, these authors proposed different ways of weighing toxicity data points to arrive at a more meaningful HC_5. One such approach (the 'enviro' approach) was to place more weight on those taxa which are well represented in the environment – e.g. there are more algal species than fish species so the former should have a greater representation in an overall SSD. However, body size is generally inversely correlated with abundance and hence, recovery of the more abundant organisms tends to be more rapid. We suggest that it would be more appropriate to use generation time as a weighting factor – thereby placing more weight on those species which are less easily recovered following a toxic insult.

Validation potential

As opposed to the terrestrial compartment, the use of SSD in risk assessment has been proposed and implemented for a longer period of time (Suter 2002). Although more complex than first-tier assessments used for product evaluation, these are more ecologically realistic (section 2.4) and many studies have assessed the concept of species protection through comparison of field data (e.g. Solomon et al 1996, Van den Brink et al 2002, Hose and Van den Brink 2004, Brock et al. 2004 are more recent examples). In Brock et al. (2004), the protective value of both the SSD approach and the first-tier approach was field validated for two photosynthesis-inhibiting herbicides. The species sensitivities were assessed using laboratory data for algae, vascular plants, aquatic invertebrates, and fish. For all taxa, HC_5 and HC_{10} values were calculated for metribuzin and HC_{10} values were calculated for metamitron. It was found that these were very similar to the ecological threshold concentrations for primary producers in the mesocosm studies and may be used to set maximum permissible concentrations in a cost effective way. Also, Brock and colleagues found that the benchmarks based on first Tier laboratory data are protective and even conservative when compared to field data. Similar results were found by Brock et al. (2000 a and b) in a major review study which compared endpoints from laboratory studies with insecticides and herbicides to the results of field studies.

By definition, first Tier criteria are meant to be conservative and are more useful at fast-tracking pesticides or pushing them into higher Tier testing than assessing the relative risk of in use products. In our estimation, extrapolation factors of 10 or 100 should always be treated with some suspicion. Of greater interest to us is the shape of the relationship between calculated TERs and the scale and duration of the actual aquatic community impact. Ideally, we would like to translate TERs into a meaningful index of aquatic community disruption analogous to the probability of field kills in birds. An on-going aspect of our NAESI project (with assistance from the USEPA for data access) is a review of available pond and mesocosm studies from a larger sample of pesticides to correlate laboratory toxicity endpoints to the extent and duration of community disturbance from a pesticide exposure.

Database needs and progress to date

As part of this project, a large database of aquatic toxicity endpoints has been assembled. As a first step, we intend to use this database to prioritize products for development of water quality criteria. A longer-term goal is to explore these new SSD concepts to arrive at a better predictor of aquatic ecosystem impact.

NEXT STEPS

Work plans:

- Circulate scoping report for comments
- Modify work plans on the basis of feedback received
- Adopt preliminary ranking system to prioritize pesticides for development of water quality objectives

Database creation:

- Complete database of pesticide use patterns and label rates
- Complete and clean aquatic endpoint database for analyses

Complete pollinator and terrestrial invertebrate database

Update plant database (C. Boutin collaborator)

Analyses:

Already underway:

- Mammal/bird, oral/dermal comparisons; mammalian modeling (Kannan Krishnan, Summer NSERC student applied for)
- Aquatic toxicity endpoints as predictors of mesocosm and pond study results (MSc underway)

Not yet begun:

- General analyses:
- Study the possibility of weighting toxicity data by generation or recovery time
- Propose strategies for missing data
- Look at formulation differences in toxicity endpoints
- Consider whether chronic toxicity indices are possible with any taxon
- Taxon-specific analyses:
- Look at combining the different bee risk indicators; validation with incident data (UK collaborators)

- Develop strategies for non-monotonic bee risk (?)
- Comparison of endpoints in bees vs. other terrestrial invertebrates. Attempt validation with beneficials database (Oregon State Univ. collaborators)
- Search literature for earthworm impact studies and use to validate index.
- Look at available plant information and develop index (collaborator C. Boutin
- Validation of fish toxicity endpoints through incident data
- Obtain mammalian field studies to establish benchmarks
- Comparison of amphibian and fish data (B. Pauli)

Needed collaboration:

PMRA for development of standard scenarios

Cessna/McQueen – if outputs to surface waters completed in time

Track European HAIR initiative

Continue obtaining reports and reference material e.g. OECD terrestrial indicator group output

Development of pesticide application factors through consultation process with experts

Table 1: List of the reviewed systems, in approximate order of publication. Numbering of systems is arbitrary.

System name	Country	Authors	System ID
Environmental Impact Quotient (EIQ)	U.S.A.	Kovach et al. (1992)[1] in Van der Werf (1996) and Levitan (1997)	2
Dutch Environmental Yardstick for Pesticides (EYP)	Netherlands	Reus (1992)[1] and Reus & Pak (1993)[1] in appendix to Reus et al. (1999); Reus and Leendertse (2000)	6
Environmental economic injury levels		Higley and Wintersteen (1992)[1] in Van der Werf (1996)	-
Adverse water quality impact		Hornsby (1992)[1] in Van der Werf (1996)	-
Stemilt Growers Integrated Fruit Production Responsible Choice Point Summary	U.S.A.	Stemilt Growers (1993)[1] in Levitan (1997)	3
Environmental Potential Risk Indicator for Pesticides (EPRIP)	Italy	Based on Del Re et al. (1993)[1] as described by Trevisan et al. in appendix to Reus et al. 1999	13
PESTDECIDE©	Australia	Penrose et al. (1995). Supersedes 'Rating Index' of Penrose et al. (1994)	4
Consumer Union's Indices of Trends in Agricultural Pesticide Risk	U.S.A.	Benbrook et al. (1996), also reviewed by Levitan (1997)	7
U. of California Environmental Health Policy Program Ranking System	U.S.A.	Pease et al. (1996)[1] in Levitan (1997)	10
CHEMS	U.S.A.	Swanson et al (1997)	-
SYNOPS	Germany	Version 1.1.: Gutsche & Rossberg (1997); Version 2 Gutsche & Rossberg (1998); Also in OECD (2000b)	9
Environmental performance indicators for pesticides (p-EMA)	U.K.	Lewis et al. 1997a&b[1] in appendix to Reus et al. 1999; Brown et al. (2003); Hart et al. (2003)	12
USDA ERS Chronic and Acute Risk Indicators of Pesticide Use	U.S.A.	Barnard (1997)[1] in Levitan (1997)	8
Ipest	France	Van der Werf and Zimmer (1998)	5
The Hasse Diagram (HD)	Denmark	In appendix to Reus et al. 1999; also in Sørensen et al. (1998)	11

Table 1: List of the reviewed systems, in approximate order of publication. Numbering of systems is arbitrary.

System name	Country	Authors	System ID
System for predicting the environmental impact of pesticides (SyPEP)	Belgium	Pussemier (In appendix to Reus et al. 1999)	14
Pesticide Environmental Risk Indicator (PERI)	Sweden	Nilsson (In appendix to Reus et al. 1999)	15
Bees Risk Indicator (BRI)	Italy	Villa et al. (2000)	16
The RIVM Indicator : AARI (and ATRI)	Netherlands	In the OECD (2000b); also Luttik, pers. Comm.[1]	17
Danish Load Index (DLI)	Denmark	Not reported[1]; discussed in OECD (2004) and also in OECD (2000b)	18
European Standard Characteristic Of non-target arthropod Regulatory Testing (ESCORT_2)	Europe	Candolfi et al. (2000)	19
Frequency of Application (FA)	Denmark	Not reported[1]; discussed in OECD (2004) and also in OECD (2000b)	20
Norwegian Indicator (NARI)	Norway	Norwegian Agricultural Inspection Services (2000)	22
Short-term Pesticide Risk Index for Hypogean Soil systems (PRIHS-1)	Italy	Finizio et al. (2001)	24
Long-term Pesticide Risk Index for Hypogean Soil systems (PRIHS-2)	Italy	Finizio et al. (2001)	24
Short-term Pesticide Risk Index for Epygean Soil systems (PRIES-1)	Italy	Finizio et al. (2001)	24
Long-term Pesticide Risk Index for Epygean Soil systems (PRIES-2)	Italy	Finizio et al. (2001)	24
Short-term Pesticide Risk Index for Surface Water Systems (PRISW-1)	Italy	Finizio et al. (2001)	24
Long-term Pesticide Risk Index for Surface Water Systems (PRISW-2)	Italy	Finizio et al. (2001)	24
Environmental Risk for pesticides (ERIP); for Hypogean or Epygean Soil Systems and Surface Water Systems	Italy	Finizio et al. (2001)	24
Multi Attribute Toxicity Factors (MATF)	U.S.A.	Benbrook et al. (2002)	21
Pesticide Occupational and Environmental Risk (POCER)	Belgium	Vercruysse and Steurbaut (2002)	23
Ecological Relative Risk (EcoRR)	Australia	Sánchez-Bayo et al. (2002)	26
Probability of bird mortality	Canada	Mineau (2002, 2004)	25

Table 1: List of the reviewed systems, in approximate order of publication. Numbering of systems is arbitrary.

System name	Country	Authors	System ID
SCRAM	U.S.A.	Mitchell et al.	29
Recommendations for Prioritizing Risk Reduction Strategies Based on Toxicity Loading by Commodity, By Region, and by Pesticide	Canada	WWF (2003); draft	28
PEI Relative Ranking System	Canada	Dunn (2004)	1
A Pesticide Priority List Evaluation Scheme: APPLES	Canada	CCME (2004); draft	27

[1] *Original publication not retrieved; information from reviews*

Table 2: Compartments and taxa included in the reviewed systems

ID System	1 PEI	2 EIQ	3 Stemilt	4 PESTDECIDE	5 Ipest	6 Dutch Yardstick	7 Cons. Union	8 ERS	9 SYNOPS	10 U. California	11 Hasse Diagram	12 p-EMA	13 EPRIP	14 SyPEP	15 PERI	16 BRI	17 AARI and ATRI	18 Load Index	19 ESCORT_2	20[5] FA	21 MATF	22 Norwegian	23 POCER	24 ERIP	25 Mineau	26 EcoRR	27 APPLES	28 WWF	29 SCRAM
Human health	X	X	X	X	X		X	X		X			X	X							X[6]	X						X	X
Terrestrial:		X	X	X		X			X	X		X	X		X	X	X	X[4]			X	X	X	X		X		X	X[9]
Mammal				X					[8]			X										X		X		X		X	
Birds		X		X					[8]	X		X					X				X		X	X	X	X		X	
Bees		X		X								X			X	X						X	X	X				X	
Other beneficials		X	X	X								X							X				X	X					
Earthworms						X			X	X[2]		X	X		X		X					X	X	X					
Micro-organisms																								X					
Amphibians																										X		X	
Terrestrial plants																													
Aquatic:	X	X			X	X			X	X	X[3]	X	X	X	X		X	X[4]			X	X	X	X		X	X	X	X[9]
Fish	X	X			X	X[1]			X	X		X	X	X[1]			X				X	X	X	X		X	X	X	
Aquatic invertebrates	X				X	X[1]			X			X	X	X[1]	X		X				X	X	X	X		X	X	X	
Algae					X				X			X	X		X		X					X	X	X			X		
Aquatic plants												X										X[7]							

[1] *Referred to as "risk to water organisms"; fish and aquatic invertebrates were assumed; algae could also be considered.*

[2] *Reported only as invertebrates; earthworm was assumed, although it could also be aquatic invertebrates*

[3] *Risk to groundwater contamination is modelled based on physico-chemical properties and usage data; no toxicological endpoints.*

[4] *Can be calculated for any organisms*

[5] *Indicator of intensity of pesticide use only*

[6] *Not clear if acute risk to mammals refers to mammals other than human; chronic risk is clearly for humans*

[7] *Reported as algae/water plants*

[8] *Synops 1.1proposed considering avian and mammalian NOECs in order to assess food chain risk*

[9] *The most sensitive value for toxicity is used, independent of the taxa or the compartment; does not appear in Tables 5 and 6*

Table 3: List of toxicity input variables used in the reviewed terrestrial indices

Terrestrial taxa	Endpoint	System ID
Mammals	LD_{50}, oral	12
Mammals	LD_{50}, dermal	4,24
Mammals	LD_{50}	26,28
Mammals	NOEL	24
Mammals	NOEL/LOEL	28
Birds	LD_{50}	10,12,21,(22)*,23,24,25
Birds	LC_{50}	2,12,22
Birds	LC/LD_{50}	26,28
Birds	NOEL	24
Birds	NOEL/LOEL	28
Bees	LD_{50}	12,23,24
Bees	LD_{50}, oral	15,16,22
Bees	LD_{50}, contact	16,22
Bees	NOEL	24
Other beneficial arthropods	LR_{50}	19
Other beneficial arthropods	NOEL	24
Earthworms	LC_{50}	6,9,10,12,13,15,22,23,24
Earthworms	EC_{50}	24
Earthworms	NOEC	6,9,24

* *Authors suggest that LD_{50} may be used instead of the LC_{50} when the latter is not available*

Table 4: List of toxicity input variables used in the reviewed aquatic indices

Aquatic taxa	Endpoint	System ID
Fish	LC_{50}	1,2,6,9,10,12,13,21,27,28
Fish	EC_{50}	5
Fish	LC/EC_{50}	17,22,26,28
Fish	NOEC	9,12
Aquatic invertebrates	LC/EC_{50}	1,15,17,22,26,28
Aquatic invertebrates	EC_{50}	5,12,27
Aquatic invertebrates	LC_{50}	6,9,13,21
Aquatic invertebrates	NOEC	9,12
Algae	EC_{50}	5,9,12,27
Algae	LC_{50}	13
Algae	LC/EC_{50}	15,17,22,26
Algae	NOEC	9
Plants	EC_{50}	12

Table 5: Incorporation of toxicity data into equations for the terrestrial compartment

a. MAMMALS

Index name	System ID	Input toxicity variables	Variable manipulation	Index manipulations
Mammalian toxicity	Ref 4: PESTDECIDE©	Mammalian dermal LD_{50}	Not reported	Dermal LD_{50} is scored Score 1: LD_{50} > 1000 mg/kg Score 2: LD_{50} = 501 – 1000 mg/kg Score 3: LD_{50} = 51 – 500 mg/kg Score 4: LD_{50} = 5 – 50 mg/kg Score 5: LD_{50} < 5 mg/kg Toxicity score added to scores for other variables
Acute risk to mammals	Ref 12: p-EMA	LD_{50}	Lowest for rat or mouse, where not available use lowest species available	TER = LD_{50} / exposure from contaminated food Exposure based on concentration in food, daily food intake and body weight. Result is scored based on threshold values
Load to mammals	Ref 18: Danish Load Index	Acute toxicity	Average, min, or max	DLI = sum for all a.i. (sales / toxicity * area of land) Calculated on a yearly basis to track changes
Short-term risk to mammals in a hypogean soil system	Ref 24: PRIHS-1	Dermal LD_{50}	Not reported	$TER_{\_short\text{-}term} = LD_{50} / PEC$ PEC based on max application rate, soil depth and bulk density Result is scored from 0-8 PRIHS-1 = [5.5 * Score(Earthworm)] + [5 * Score(Beneficial)] + [2 * Score(Mammal)] For ERIP, all compartments are added but scoring and weights in each compartments change

a. MAMMALS

Index name	System ID	Input toxicity variables	Variable manipulation	Index manipulations
Long-term risk to mammals in a hypogean soil system	Ref 24: PRIHS-2	2-yr NOEL	Not reported	$TER_{long\ term}$ = NOEL / (Bioconcentration factor * $PEC_{short\ term}$) $PEC_{short\ term}$ calculated as above Result is scored from 0-8 PRIHS-2 = [4 * Score(Earthworms)] + [4 * Score(Micro-organisms)] + [3 * Score(Arthropods)] + [1.5 * Score(Mammal)] For ERIP, all compartments are added but scoring and weights in each compartments change
Short-term risk to mammals in a epygean soil system	Ref 24: PRIES-1	LD_{50}	Not reported	$TER_{short\ term}$ = LD_{50} / Total Daily Intake Result is scored from 0-8 PRIES-1 = [3 * Score(Bees)] + [4 * Score(Birds)] + [3 * Score(Beneficial)] + [2.5 * Score(Mammal)] For ERIP, all compartments are added but scoring and weights in each compartments change
Long-term effect on mammals in a epygean soil system	Ref 24: PRIES-2	NOEL	Not reported	NOEL is scored from 0-4 PRIES-2 = (Sum of all 5 effect scores / 5) * [Score(Air + Soil affinity) / 2] * Score(Bioaccumulation) * Score (Persistence) * Score (Max application rate) For ERIP, all compartments are added but scoring and weights in each compartments change

a. MAMMALS

Index name	System ID	Input toxicity variables	Variable manipulation	Index manipulations
Ecotoxicity; terrestrial compartment	Ref 26: EcoRR	LC/LD_{50}; oral or dermal depending on compartment considered	Geometric mean for a given taxon	$\frac{\text{Sum of } [(\text{toxicity geomean})_{taxon} / (S_{taxon}/N)]}{N}$ S_{taxon} is the number of species in one of the taxa considered for a given compartment and N is the total number of species of all taxa considered in that compartment Then toxicity/exposure ratio where exposure specific to compartment; based on application rate, its partitioning into a given compartment, degradation rate, and BCF
Toxicity score	Ref 28: WWF	LC/LD_{50}	Not reported	Score based on Kovach's EIQ

b: BIRDS

Index Name	System ID	Input toxicity variables	Variable manipulation	Index manipulations
Impact on birds	Ref 2: Environmental Impact Quotient	Avian LC_{50}	Not reported	LC_{50} is scored: Score 1: > 1000 ppm Score 3: 100 – 1000 ppm Score 5: 1 – 100 ppm Combined with scored for half-life on plants and soil: 3 * Score(Bird Toxicity) * $\frac{[\text{Score(Soil half-life)} + \text{Score(Plant half-life)}]}{2}$
Ecological Health	Ref 10: U. of California	Avian LD_{50}	Not reported	LD_{50} is scored from 1 to 4; criteria not reported Ecological Health = Score(Avian) + Score(Invertebrate) + Score(Fish) + Score(Bioconcentration Factor)
Acute risk to birds	Ref 12: p-EMA	Oral LD_{50}	Lowest of bobwhite quail, Japanese quail or mallard; where not available use lowest other species; at the end of computations, use the lowest between acute and short-term risk for final index for birds	TER = LD_{50} / exposure from contaminated food Exposure based on concentration in food, daily food intake and body weight. Result is scored based on threshold values
Short-term risk to birds	Ref 12: p-EMA	5-d dietary LC_{50}	As above	TER = LC_{50} / concentration in food Result is scored based on threshold values

b: BIRDS

Index Name	System ID	Input toxicity variables	Variable manipulation	Index manipulations
Acute risk to terrestrial wildlife	Ref 17: ATRI	Partridge acute toxicity	Geometric mean of available data?	Assumed that equation is same as for aquatic : ATRI = sum for all active ingredients of (PEC/TOX) * area weighted average
Load to birds	Ref 18: Danish Load Index	Acute toxicity	Average, min, or max	DLI = sum for all a.i. (sales / toxicity * area of land) Calculated on a yearly basis to track changes
Ecological Toxicity (avian index)	Ref 21: MATF	Bird LD_{50} [ECOTOX]	Not reported	Avian index = $1/LD_{50\ bird}$ * scaling factor; different scaling factors are used, depending on the pesticide, to narrow the wide range of index values. ECO = Daphnia index + Fish index + Avian index
Risk to terrestrial organisms (T)	Ref 22: Norwegian Indicator	Bird LC_{50} (LD_{50} if LC_{50} not available)	Not reported	$TER_{bird} = LC_{50} / PEC_{food}$ PEC_{food} = application rate * RUD (can use the acute oral LD_{50} value, but must take into account the quantity of contaminated food ingested) Score 0: TER >10 Score 2: TER = 1-10 Score 4: TER < 1 T = the highest of scores for earthworms, bees, or birds

b: BIRDS

Index Name	System ID	Input toxicity variables	Variable manipulation	Index manipulations
Acute Risk to birds	Ref 23: POCER	LD_{50}	Not reported	Risk Index $_{birds}$ = (PEC_{bird} * 10) / (LD_{50} * body weight) Where10 is the criteria set by the EU Uniform Principles; PEC based on application rate, body weight. For granules, based on granule weight and % a.i. in the granule.
Short-term risk to birds	Ref 24: PRIES-1	LD_{50}	Not reported	$TER_{bird\_short\ term}$ = LD_{50} / Total Daily Intake Result is scored from 0-8 PRIES-1 = [3 * Score(Bees)] + [4 * Score(Birds)] + [3 * Score(Beneficial)] + [2.5 * Score(Mammal)] For ERIP, all compartments are added but scoring and weights in each compartments change
Long-term effect on birds	Ref 24: PRIES-2	NOEL	Not reported	NOEL is scored from 0-4 PRIES-2 = (Sum of all 5 effect scores / 5) * [Score(Air + Soil affinity) / 2] * Score(Bioaccumulation) * Score (Persistence) * Score (Max application rate) For ERIP, all compartments are added but scoring and weights in each compartments change
Probability of bird mortality	Ref 25: Mineau	Bird oral LD_{50} Rat oral LD_{50} Rat dermal LD_{50}	Geometric mean of avian toxicity	Physico-chemical data and rat data to estimate pesticide ability to penetrate skin; HD5 derived from avian data. Model result (i.e. probability of mortality) is multiplied with ‘use pattern correction factors’ according to different application scenarios

b: BIRDS

Index Name	System ID	Input toxicity variables	Variable manipulation	Index manipulations
Ecotoxicity; terrestrial compartment	Ref 26: EcoRR	LC/LD_{50}; oral or dermal depending on compartment considered	Geometric mean for a given taxon	$\dfrac{\text{Sum of }[(\text{toxicity geomean})_{taxon} / (S_{taxon}/N)]}{N}$ S_{taxon} is the number of species in one of the taxa considered for a given compartment and N is the total number of species of all taxa considered in that compartment. Then toxicity/exposure ratio where exposure specific to compartment; based on application rate, its partitioning into a given compartment, degradation rate, and BCF
Toxicity score	Ref 28: WWF	LC/LD_{50}	Not reported	Score based on Kovach's EIQ

c. BEES

Index name	System ID	Input toxicity variables	Variable manipulation	Index manipulations
Impact on bees	Ref 2: Environmental Impact Quotient	Lethality to honey bees at field doses	None; categorical variable	Lethality is scored Score 1: relatively non toxic Score 3: moderately toxic Score 5: highly toxic Combined with scored for half-life on plants Score(Lethality to honey bees) * Score(Plant half-life) * 3
Acute risk to bees	Ref 12: p-EMA	48-h LD_{50}	Lowest of oral and contact	For spray: HQ = application rate /LD_{50} HQ = 0 for seed treatments, granules and pellets, when application between Oct. and Feb. and on non-flowering crops. Result is scored based on threshold values
Acute toxicity to indicator organisms	Ref 15: PERI	Oral LD_{50}	Not reported - although an average of scores is later calculated (see Index manipulations)	If available, bee toxicity is scored: Score 1: > 100 Score 2: 10 – 100 Score 3: 1 – 10 Score 4: 0.1 – 1 Score 5: > 0.1 An average of the toxicity scores for available species (regardless of taxa) is calculated

c. BEES

Index name	System ID	Input toxicity variables	Variable manipulation	Index manipulations
Bees Risk Indicator - ingestion	Ref 16: BRI	Bee oral LD_{50}	Not reported	TER ingestion = LD_{50_oral} / Pollen PEC PEC based on octanol-air partition coeff. and persistence of chemical, as well as amount of pollen ingested
Bees Risk Indicator - contact	Ref 16: BRI	Bee contact LD_{50}	Not reported	TER contact = $LD_{50\_contact}$ / Pollen PEC PEC based on octanol-air partition coeff. and persistence of chemical, as well as amount of pollen picked up
Risk to terrestrial organisms (T)	Ref 22: Norwegian Indicator	Contact or oral LD_{50}	Not reported	HQ_{bees} = application rate / LD_{50} The highest HQ (i.e. chose between the HQ calculated with contact toxicity or with oral toxicity) is scored. Score 0: HQ <50 Score 0.5: HQ = 50 – 100 Score 1: HQ = 100 – 1000 Score 1.5: HQ = 1000 – 10000 Score 2: HQ > 10000 T = the highest of scores for earthworms, bees, or birds
Acute Risk to bees	Ref 23: POCER	LD_{50}	Use the minimum between the oral and contact LD_{50} value	Risk Index $_{bees}$ = application rate / (LD_{50} * 50) Where 50 is the criteria set by Uniform Principles

c. BEES

Index name	System ID	Input toxicity variables	Variable manipulation	Index manipulations
Short-term risk to bees	Ref 24: PRIES-1	LD_{50}	Not reported	HQ = maximum rate of application / LD_{50} Result is scored from 0-8 PRIES-1 = [3 * Score(Bees)] + [4 * Score(Birds)] + [3 * Score(Beneficial)] + [2.5 * Score(Mammal)] For ERIP, all compartments are added but scoring and weights in each compartments change
Long-term effect on bees	Ref 24: PRIES-2	NOEL	Not reported	NOEL is scored from 0-4 PRIES-2 = (Sum of all 5 effect scores / 5) * [Score(Air + Soil affinity) / 2] * Score(Bioaccumulation) * Score (Persistence) * Score (Max application rate) For ERIP, all compartments are added but scoring and weights in each compartments change
Toxicity score	Ref 28: WWF	LC/LD_{50}	Not reported	Score based on Kovach's EIQ

d. OTHER BENEFICIAL ARTHROPODS

Index name	System ID	Input toxicity variables	Variable manipulation	Index manipulations
Impact on beneficials	Ref 2: Environmental Impact Quotient	Beneficial arthropod toxicity	None; categorical variable	Beneficial arthropod toxicity is scored Score 1: low impact Score 3: moderate impact or post-emergent herbicides Score 5: severe impact Combined with scored for half-life on plants Score(Beneficial arthropod toxicity)*Score(Plant half-life)*5
Risk to non-target arthropods	Ref 12: p-EMA	Not a toxicity endpoint; insecticidal activity of product	None; categorical variable	Score 0: no insecticidal activity or solid formulations or seeds Score -50: Selective insecticides or ICP insecticides Score -90: Active against a broad spectrum of insects Score -80: Other insecticides or other pesticides with insecticidal activity
Risk to non-target arthropods – in field	Ref 19: ESCORT_2	LR_{50} For indicator sp. Aphidius rhopalosiphi and Typhlodromus pyri	Not reported	HQ_{in_field} = (application rate * MAF) / LR_{50} Where LR_{50} is the application rate causing 50% mortality ; Multiple Application Factor derived from the half-life of the product, the spray interval and the number of applications

d. OTHER BENEFICIAL ARTHROPODS

Index name	System ID	Input toxicity variables	Variable manipulation	Index manipulations
Risk to non-target arthropods – off field	Ref 19: ESCORT_2	LR_{50} For indicator sp. Aphidius rhopalosiphi and Typhlodromus pyri	Not reported	HQ_{off_field} = [application rate * MAF * (drift factor / vegetation distribution factor) / LR_{50}] * safety factor Where LR_{50} is the application rate causing 50% mortality ; Multiple Application Factor derived from the half-life of the product, the spray interval and the number of applications; safety factor of 10 to account for the extrapolation from the indicator species used in first tier to all other off-field non-target arthropods
Acute risk to beneficial arthropods	Ref 23: POCER	% reduction in control capacity [online Koppert database at http://www.koppert.nl/e0110.html and Biobest database]	Use the sp. with the highest % reduction in control capacity (calculate arithmetic average when range)	Risk Index $_{beneficial\ arthropods}$ = (% reduction of control capacity – 25) / (100 – 25) Risk is null if % reduction in control capacity is less than 25%. Risk is null if pesticide applied using a method other than spray. Risk for herbicides considered to be null.

d. OTHER BENEFICIAL ARTHROPODS

Index name	System ID	Input toxicity variables	Variable manipulation	Index manipulations
Short-term risk to beneficial arthropods	Ref 24: PRIHS-1 and PRIES-1	% inhibition	Not reported	Score 0: % inhibition is null at (2 * max application rate) Score 2: % inhibition between 0% - 30% at the max application rate Score 4: % inhibition is > 30% at the max application rate Score 8: % inhibition is > 30% at (0.5 * max application rate) PRIHS-1 = [5.5 * Score(Earthworm)] + [5 * Score(Beneficial)] + [2 * Score(Mammal)] PRIES-1 = [3 * Score(Bees)] + [4 * Score(Birds)] + [3 * Score(Beneficial)] + [2.5 * Score(Mammal)] For ERIP, all compartments are added but scoring and weights in each compartments change
Long-term risk to beneficial arthropods	Ref 24: PRIHS-2	% inhibition	Not reported	Scoring same as for short-term risk PRIHS-2 = [4 * Score(Earthworms)] + [4 * Score(Micro-organisms)] + [3 * Score(Arthropods)] + [1.5 * Score(Mammal)] For ERIP, all compartments are added but scoring and weights in each compartments change

d. OTHER BENEFICIAL ARTHROPODS

Index name	System ID	Input toxicity variables	Variable manipulation	Index manipulations
Long-term effect on beneficial arthropods	Ref 24: PRIES-2	NOEL	Not reported	NOEL is scored from 0-4 PRIES-2 = (Sum of all 5 effect scores / 5) * [Score(Air + Soil affinity) / 2] * Score(Bioaccumulation) * Score (Persistence) * Score (Max application rate) For ERIP, all compartments are added but scoring and weights in each compartments change

e. EARTHWORMS

Index name	System ID	Input toxicity variables	Variable manipulation	Index manipulations
Risk to soil organisms – short-term	Ref 6: Dutch Environmental Yardstick	LC_{50}; assumed earthworm [data from industry; for product registration]	Not reported	Environmental impact points = $100*PEC_{soil\_direct}$ / $(0.1*LC_{50})$ Where PEC based on degradation rate in soil, mobility in soil, and dose applied
Risk to soil organisms – long-term	Ref 6: Dutch Environmental Yardstick	NOEC; assumed earthworm [data from industry; for registration]	Not reported	Environmental impact points = $100*PEC_{soil\_after\_two\_years}$ / (0.1*NOEC) Where PEC based on degradation rate in soil, mobility in soil, and dose applied
Risk to earthworms - acute	Ref 9: SYNOPS_2	Earthworm LC_{50}	Not reported	$PEC_{soil\ short\text{-}term}$ / LC_{50} PEC based on application rate, drift and crop interception
Risk to earthworms – long-term	Ref 9: SYNOPS_2	Earthworm NOEC	Not reported	$PEC_{soil\ long\text{-}term}$ * test duration / NOEC PEC is a function of degradation rate and pesticide absorption to soil; calculated over a one year span
Ecological Health	Ref 10: U. of California	Invertebrate LC_{50}; assumed earthworm	Not reported	LC_{50} is scored from 1 to 4; criteria not reported Terrestrial invertebrate (rather than aquatic) is assumed Ecological Health = Score(Avian) + Score(Invertebrate) + Score(Fish) + Score(Bioconcentration Factor)
Acute risk to earthworms	Ref 12: p-EMA	14-d LC_{50}	Eisenia foetida or lowest available species	TER = LC_{50} / initial soil concentration Result is scored; soil concentration obtained through modelling
Risk to earthworms	Ref 13: EPRIP	Earthworm LC_{50}	Not reported	PEC_{soil} / LC_{50} After a single application:

e. EARTHWORMS

Index name	System ID	Input toxicity variables	Variable manipulation	Index manipulations
				PEC_{soil} = aprate * (1 - f_{int}) / (100 * mixing depth * soil bulk density) For many applications: PEC_n = PEC_{soil} * (1 – exp^{-nki}) / (1- exp^{-ki}) Where n is the number of applications; k is the dissipation rate = ln2/soil half-life; i is the number of days
Acute toxicity to indicator organisms	Ref 15: PERI	Earthworm LC_{50}	Not reported - although an average of scores is later calculated (see Index manipulations)	If available, earthworm LC_{50} is scored: Score 1: > 1000, Score 2: 1000 – 100, Score 3: 10 – 100, Score 4: 1 – 10, Score 5: < 1 An average of toxicity scores for available species. is calculated
Acute risk to earthworms	Ref 17: ATRI	Earthworm acute toxicity	Geometric mean of available data?	Assumed that equation is same as for aquatic : ATRI = sum for all active ingredients of (PEC/TOX) * area weighted average
Risk to terrestrial organisms (T)	Ref 22: Norwegian Indicator	Earthworm 14-day LC_{50}	Not reported	$TER_{earthworm}$ = LC_{50} / PEC_{soil} PEC_{soil} = aprate * (1 - interception) / (100 * mixing depth * soil bulk density) Score 0: TER > 100 Score 2: TER = 10-100 Score 3: TER < 10 T = the highest of scores for earthworms, bees, or birds

e. EARTHWORMS

Index name	System ID	Input toxicity variables	Variable manipulation	Index manipulations
Acute Risk to earthworms	Ref 23: POCER	Earthworm LC_{50}	Not reported	Risk Index $_{earthworms}$ = (PIEC * 10) / LC_{50} Where 10 is the criteria set by Uniform Principles; PEC based on application rate, fraction reaching soil, depth and density of soil.
Short-term risk to earthworms	Ref 24: PRIHS-1	EC_{50}	Not reported	$TER_{short\text{-}term} = EC_{50} / PEC$ PEC based on max application rate, soil depth and bulk density Result is scored from 0-8 PRIHS-1 = [5.5 * Score(Earthworm)] + [5 * Score(Beneficial)] + [2 * Score(Mammal)] For ERIP, all compartments are added but scoring and weights in each compartments change
Long-term risk to earthworms	Ref 24: PRIHS-2	14-day NOEC	Not reported	$TER_{long\ term} = PEC_{short\text{-}term} * ((1 - e^{-kt}) / kt)$ $PEC_{short\text{-}term}$ calculated as above t is time of the toxicity test; $k = 2/DT_{50}$ Result is scored from 0-8 PRIHS-2 = [4 * Score(Earthworms)] + [4 * Score(Micro-organisms)] + [3 * Score(Arthropods)] + [1.5 * Score(Mammal)] For ERIP, all compartments are added but scoring and weights in each compartments change

f. SOIL MICRO-ORGANISMS

Index name	System ID	Input toxicity variables	Variable manipulation	Index manipulations
Long-term risk to micro-organisms	Ref 24: PRIHS-2	% Inhibition of activity	Not reported	Score 0: % inhibition is null at (2 * max application rate) Score 2: % inhibition is between 0% - 25% at the max application rate Score 4: % inhibition is > 25% at the max application rate Score 8: % inhibition is > 25% at (0.5 * max application rate) PRIHS-2 = [4 * Score(Earthworms)] + [4 * Score(Micro-organisms)] + [3 * Score(Arthropods)] + [1.5 * Score(Mammal)] For ERIP, all compartments are added but scoring and weights in each compartments change

g. TERRESTRIAL PLANTS

Index name	System ID	Input toxicity variables	Variable manipulation	Index manipulations
Long-term effect on plants	Ref 24: PRIES-2	Phytotoxicity	None; categorical	Score 0: - phytotoxic Score 4: + phytotoxic PRIES-2 = (Sum of all 5 effect scores / 5) * [Score(Air + Soil affinity) / 2] * Score(Bioaccumulation) * Score (Persistence) * Score (Max application rate) For ERIP, all compartments are added but scoring and weights in each compartments change

Table 6: Incorporation of toxicity data into equations for the aquatic compartment

a. FISH

Index name	System ID	Input toxicity variables	Variable manipulation	Index manipulations
Environmental Hazard	Ref 1: Environment Canada's modified CHEMS	Preferential use of rainbow trout 96-hour LC_{50} [ECOTOX;PM]	Geometric mean if more than one value; tiered approach for missing data	LC_{50} is scored Score 0: ≥ 1000 mg/L; Score 5: < 1 mg/L ; Between cut-off values, Score = $-1.67\log LC_{50} + 5.0$ The score for fish is then summed with the score for Daphnia (below)
Impact on aquatic vertebrates	Ref 2: Environmental Impact Quotient	Fish 96-hour LC_{50}	Not reported	LC_{50} is scored Score 1: > 10 ppm ; Score 3: 1 – 10 ppm; Score 5: < 1 ppm Score(Fish toxicity) * Score(Surface runoff potential)
Risk of surface water contamination	Ref 5: Ipest	Fish EC_{50}	Chose the most sensitive organism between algae, Daphnia, and fish	Fuzzy logic If fish is the most sensitive organism, $\log_{10}EC_{50}$ will be considered for he favourable subset if > 2 (100 mg/L); unfavourable if < - 2 (0.01mg/L). Combined with runoff and drift potential, and position of application.
Risk to water organisms (surface water)	Ref 6: Dutch Environmental Yardstick	Fish LC_{50} [registration data]	Chose the most sensitive organism	If fish is the most sensitive organism: Environmental impact points = $100*PEC/LC_{50water\_organism}$ Where PEC based on the method of application and dose applied

a. FISH

Index name	System ID	Input toxicity variables	Variable manipulation	Index manipulations
Risk to fish - acute	Ref 9: SYNOPS_2	Fish LC_{50}	Not reported	$PEC_{water\ short\text{-}term} / LC_{50}$ PEC based on drift and the proportion of field lengths that border a water body in a region
Risk to fish – long-term	Ref 9: SYNOPS_2	Fish NOEC	Not reported	$PEC_{water\ long\text{-}term}$ * test duration / NOEC PEC is a function of degradation rate and pesticide absorption to soil; calculated over a one year span
Ecological Health	Ref 10: U. of California	Fish LC_{50}	Not reported	Fish LC_{50} is scored from 1 to 4; criteria not reported Ecological Health = Score(Avian) + Score(Invertebrate) + Score(Fish) + Score(Bioconcentration Factor)
Acute risk to fish	Ref 12: p-EMA	96-hour LC_{50}	Rainbow trout; if n.a. use (1) bluegill sunfish, (2) lowest other fish	TER = acute toxicity/PEC PEC based on Application rate; Soil type; Soil organic matter; Crop cover; Method of application
Chronic risk to fish	Ref 12: p-EMA	21-day NOEL	Rainbow trout; if n.a. use (1) bluegill sunfish, (2) lowest other fish	TER = chronic toxicity/PEC PEC based on Application rate; Soil type; Soil organic matter; Crop cover; Method of application
Risk to fish – drift	Ref 13: EPRIP	Fish LC_{50}	Not reported	PEC_{drift} / LC_{50} PEC_{drift} = aprate $* f_{drift}$ / volume of water in the ditch Result is scored from 1-5; the max score (via drift or runoff) is incorporated into the system

a. FISH

Index name	System ID	Input toxicity variables	Variable manipulation	Index manipulations
Risk to fish – runoff	Ref 13: EPRIP	Fish LC_{50}	Not reported	PEC_{runoff} / LC_{50}; depends on the slope, soil texture, intensity of the rain event, the distance between the treated area and the ditch, and on the elapsed time between pesticide application and onset of rainfall Result is scored from 1-5; the max score (via drift or runoff) is incorporated into the system
Acute risk to fish	Ref 17: AARI	Fish LC/EC_{50}	Geometric mean of available data	AARI = sum for all active ingredients of (PEC/TOX) * area weighted average PEC = PECditch = mean dosage (kg/ha) * 0.4 * mean fraction drift; based on a model calculation TOX is acute toxicity to aquatic organisms Area-weighted average = kg a.i. sold / mean dosage
Load to fish	Ref 18: Danish Load Index	Acute toxicity	Average, min, or max	DLI = sum for all a.i. (sales / toxicity * area of land) Calculated on a yearly basis to track changes
Ecological Toxicity (fish index)	Ref 21: MATF	Rainbow trout LC_{50} Bluegill LC_{50} [ECOTOX]	Not reported	Fish index = Average of [(1/rainbow trout LC_{50}) * scaling factor] and [(1/bluegill LC_{50}) * scaling factor] Different scaling factors are used, depending on the pesticide, to narrow the wide range of index values. ECO = Daphnia index + Fish index + Avian index

a. FISH

Index name	System ID	Input toxicity variables	Variable manipulation	Index manipulations
Risk to aquatic organisms (A)	Ref 22: Norwegian Indicator	Fish LC/EC_{50}	Not reported	$TER = LC/EC_{50} / PEC$ PEC is based on spray drift and surface runoff For acute studies with invertebrates or fish: Score 0: TER > 100; Score 1: TER = 10 – 100 Score 2: TER = 1 – 10; Score 3: TER = 0.1 – 1 Score 4: TER < 0.1
Acute risk to water organism	Ref 23: POCER	Fish LC_{50}	Chose the lowest of these: LC_{50} for fish/100 EC_{50} for Daphnia/100 NOEC for algae/10	PEC/chosen toxicity PEC based on application rate, drift, width and depth of ditch
Short-term risk to fish	Ref 24: PRISW-1	Fish LC_{50}	Not reported	$TER_{fish} = LC_{50} / PEC_{short\text{-}term}$ PEC based on drift and runoff Result is scored from 0-8 PRISW-1 = [3 * Score(Algae)] + [4 * Score(Daphnia)] + [5.5 * Score(Fish)] For ERIP, all compartments are added but scoring and weights in each compartments change

a. FISH

Index name	System ID	Input toxicity variables	Variable manipulation	Index manipulations
Long-term risk to fish	Ref 24: PRISW-2	Fish NOEL	Not reported	TER_{fish} = NOEL/Theoretical Exposure in Water TEW based on fugacity model results, application rate, and persistence Result is scored from 0-8 PRISW-2 = [2 * Score(Algae) + 3 * Score(Daphnia) + 3 * Score(Fish)] * B * S Where B is the bioaccumulation potential (K_{ow}) S is the % distribution of substance in sediment (fugacity level 1) For ERIP, all compartments are added but scoring and weights in each compartments change
Ecotoxicity; aquatic compartment	Ref 26: EcoRR	LC_{50}	Geometric mean for a given taxon; oral or dermal depending on compartment considered	$\dfrac{\text{Sum of } [(\text{toxicity geomean})_{taxon} / (S_{taxon}/N)]}{N}$ S_{taxon} is the number of species in one of the taxa considered for a given compartment and N is the total number of species of all taxa considered in that compartment. Then toxicity/exposure ratio where exposure specific to compartment; based on application rate, its partitioning into a given compartment, degradation rate, and BCF

a. FISH

Index name	System ID	Input toxicity variables	Variable manipulation	Index manipulations
Aquatic toxicity	Ref 27: APPLES	Preferential use of rainbow trout 96-hour LC_{50}	Most sensitive amongst fish, invertebrate or algae species	Score 1: $EC_{50}/LC_{50} > 100$; Score 4: $EC_{50}/LC_{50} = 10 – 100$ Score 7: $EC_{50}/LC_{50} = 1 – 10$; Score 10: $EC_{50}/LC_{50} = 0.1 – 1$ Score 13: $EC_{50}/LC_{50} = 0.01 – 0.1$; Score 16: < 0.01
Toxicity score	Ref 28: WWF	LC_{50}	Not reported	Score based on Kovach's EIQ

b. AQUATIC INVERTEBRATES

Index name	System ID	Input toxicity variables	Variable manipulation	Index manipulations
Environmental Hazard	Ref 1: Environment Canada's modified CHEMS	Preferential use of Daphnia magna 48-hour LC/EC_{50} [ECOTOX;PM]	Geometric mean if more than one value	Toxicity is scored (from 0 to 5, as for fish, above) The score for Daphnia is then summed with the score for fish (above)
Risk of surface water contamination	Ref 5: Ipest	Daphnia EC_{50}	Chose the most sensitive organism between algae, Daphnia, and fish	Fuzzy logic If Daphnia is the most sensitive org., $Log_{10}EC_{50}$ will be considered for he favourable subset if > 2 (100mg/L); unfavourable if < - 2 (0.01mg/L). Combined with runoff and drift potential, and position of application.
Risk to water organisms (surface water)	Ref 6: Dutch Environmental Yardstick	Invertebrate LC_{50} (registration data)	Chose the most sensitive organism	If an invertebrate the most sensitive organism: Environmental impact points = $100*PEC/LC_{50water\_organism}$ Where PEC based on the method of application and dose applied
Risk to aquatic invertebrates - acute	Ref 9: SYNOPS_2	Daphnia LC_{50}	Not reported	$PEC_{water\ short\text{-}term} / LC_{50}$ PEC based on drift and the proportion of field lengths that border a water body in a region
Risk to aquatic invertebrates – long-term	Ref 9: SYNOPS_2	Daphnia NOEC	Not reported	$PEC_{water\ long\text{-}term}$ * test duration / NOEC PEC is a function of degradation rate and pesticide absorption to soil; calculated over a one year span

b. AQUATIC INVERTEBRATES

Index name	**System ID**	**Input toxicity variables**	**Variable manipulation**	**Index manipulations**
Acute risk to Daphnia	Ref 12: p-EMA	Daphnia 48-hour EC_{50}	If Daphnia not available, use lowest other species	TER = acute toxicity/PEC PEC based on Application rate; Soil type; Soil organic matter; Crop cover; Method of application
Chronic risk to Daphnia	Ref 12: p-EMA	Daphnia 21-day NOEC	If Daphnia not available, use lowest other species	TER = chronic toxicity/PEC PEC based on Application rate; Soil type; Soil organic matter; Crop cover; Method of application
Risk to crustaceans – drift	Ref 13: EPRIP	Crustacean LC_{50}	Not reported	PEC_{drift} / LC_{50} PEC_{drift} = aprate *f_{drift} / volume of water in the ditch Result is scored from 1-5; the max score (via drift or runoff) is incorporated into the system
Risk to crustaceans – runoff	Ref 13: EPRIP	Crustacean LC_{50}	Not reported	PEC_{runoff} / LC_{50}; depends on the slope, soil texture, intensity of the rain event, the distance between the treated area and the ditch, and on the elapsed time between pesticide application and onset of rainfall Result is scored from 1-5; the max score (via drift or runoff) is incorporated into the system
Acute toxicity to indicator organisms	Ref 15: PERI	Daphnia LC/EC_{50}	Not reported - although an average of scores is later calculated (see Index manipulations)	If available, Daphnia toxicity is scored: Score 1: > 100 Score 2: 10 – 100 Score 3: 1 – 10 Score 4: 0.1 – 1 Score 5: > 0.1

b. AQUATIC INVERTEBRATES

Index name	System ID	Input toxicity variables	Variable manipulation	Index manipulations
				An average of the toxicity scores for available species is calculated
Acute risk to Invertebrates	Ref 17: AARI	Daphnia LC/EC_{50}	Geometric mean of available data	AARI = sum for all active ingredients of (PEC/TOX) * area weighted average PEC = PECditch = mean dosage (kg/ha) * 0.4 * mean fraction drift; based on a model calculation TOX is acute toxicity to aquatic organisms Area-weighted average = kg a.i. sold / mean dosage
Load to mammals	Ref 18: Danish Load Index	Acute toxicity	Average, min, or max	DLI = sum for all a.i. (sales / toxicity * area of land) Calculated on a yearly basis to track changes
Ecological Toxicity (Invertebrate index)	Ref 21: MATF	Daphnia LC_{50} [ECOTOX]	Not reported	Daphnia index = $1/LC_{50}$ * scaling factor; different scaling factors are used, depending on the pesticide, to narrow the wide range of index values. ECO = Daphnia index + Fish index + Avian index
Risk to aquatic organisms (A)	Ref 22: Norwegian Indicator	Daphnia LC/EC_{50}	Not reported	TER = LC/EC_{50} / PEC PEC is based on spray drift and surface runoff For acute studies with invertebrates or fish: Score 0: TER > 100 Score 1: TER = 10 – 100 Score 2: TER = 1 – 10 Score 3: TER = 0.1 – 1 Score 4: TER < 0.1

b. AQUATIC INVERTEBRATES

Index name	System ID	Input toxicity variables	Variable manipulation	Index manipulations
Acute risk to water organism	Ref 23: POCER	Daphnia EC_{50}	Chose the lowest of these: LC_{50} for fish/100 EC_{50} for Daphnia/100 NOEC for algae/10	PEC/chosen toxicity PEC based on application rate, drift, width and depth of ditch
Short-term risk to Daphnia	Ref 24: PRISW-1	Daphnia EC_{50}	Not reported	$TER_{Daphnia} = EC_{50} / PEC_{short\text{-}term}$ PEC based on drift and runoff Result is scored from 0-8 PRISW-1 = [3 * Score(Algae)] + [4 * Score(Daphnia)] + [5.5 * Score(Fish)] For ERIP, all compartments are added but scoring and weights in each compartments change
Long-term risk to Daphnia	Ref 24: PRISW-2	Daphnia NOEL	Not reported	$TER_{Daphnia}$ = NOEL/Theoretical Exposure in Water TEW based on fugacity model results, application rate, and persistence Result is scored from 0-8 PRISW-2 = [2 * Score(Algae) + 3 * Score(Daphnia) + 3 * Score(Fish)] * B * S Where B is the bioaccumulation potential (K_{ow}) S is the % distribution of substance in sediment (fugacity level 1) For ERIP, all compartments are added but scoring and weights in each compartments change

b. AQUATIC INVERTEBRATES

Index name	System ID	Input toxicity variables	Variable manipulation	Index manipulations
Ecotoxicity; aquatic compartment	Ref 26: EcoRR	LC_{50}	Geometric mean for a given taxon; oral or dermal depending on compartment considered	Sum of $\frac{[(\text{toxicity geomean})_{taxon} / (S_{taxon}/N)]}{N}$ S_{taxon} is the number of species in one of the taxa considered for a given compartment and N is the total number of species of all taxa considered in that compartment. Then toxicity/exposure ratio where exposure specific to compartment; based on application rate, its partitioning into a given compartment, degradation rate, and BCF
Aquatic toxicity	Ref 27: APPLES	Preferential use of 48-hour Daphnia magna EC_{50}	Most sensitive amongst fish, invertebrate or algae species	Score 1: $EC_{50}/LC_{50} > 100$ Score 4: $EC_{50}/LC_{50} = 10 – 100$ Score 7: $EC_{50}/LC_{50} = 1 – 10$ Score 10: $EC_{50}/LC_{50} = 0.1 – 1$ Score 13: $EC_{50}/LC_{50} = 0.01 – 0.1$ Score 16: $EC_{50}/LC_{50} < 0.01$
Toxicity score	Ref 28: WWF	LC_{50}	Not reported	Score based on Kovach's EIQ

c. ALGAE

Index name	System ID	Input toxicity variables	Variable manipulation	Index manipulations
Risk of surface water contamination	Ref 5: Ipest	Algae EC_{50}	Chose the most sensitive organism between algae, Daphnia, and fish	Fuzzy logic If algae is the most sensitive organism, $Log_{10}EC_{50}$ will be considered for he favourable subset if > 2 (100mg/L) ; unfavourable if < - 2 (0.01mg/L) Combined with runoff and drift potential, and position of application.
Risk to algae - acute	Ref 9: SYNOPS_2	Algae LC_{50}	Not reported	$PEC_{water\ short\text{-}term} / LC_{50}$ PEC based on drift and the proportion of field lengths that border a water body in a region
Risk to algae – long-term	Ref 9: SYNOPS_2	Algae NOEC	Not reported	$PEC_{water\ long\text{-}term}$ * test duration / NOEC PEC is a function of degradation rate and pesticide absorption to soil; calculated over a one year span
Acute risk to algae	Ref 12: p-EMA	96-hour ErC_{50}	Lowest species	TER = acute toxicity/PEC PEC based on Application rate; Soil type; Soil organic matter; Crop cover; Method of application
Risk to algae – drift	Ref 13: EPRIP	Algae LC_{50}	Not reported	PEC_{drift} / LC_{50} PEC_{drift} = aprate $*f_{drift}$ / volume of water in the ditch Result is scored from 1-5; the max score (via drift or runoff) is incorporated into the system

c. ALGAE

Index name	System ID	Input toxicity variables	Variable manipulation	Index manipulations
Risk to algae – runoff	Ref 13: EPRIP	Algae LC_{50}	Not reported	PEC_{runoff}/ LC_{50}; depends on the slope, soil texture, intensity of the rain event, the distance between the treated area and the ditch, and on the elapsed time between pesticide application and onset of rainfall Result is scored from 1-5; the max score (via drift or runoff) is incorporated into the system
Acute toxicity to indicator organisms	Ref 15: PERI	Scenedesmus, or Chorella LC/EC_{50}	Not reported - although an average of scores is later calculated (see Index manipulations)	If available, toxicity is scored: Score 1: > 100 Score 2: 10 – 100 Score 3: 1 – 10 Score 4: 0.1 – 1 Score 5: > 0.1 An average of the toxicity scores for available species is calculated
Acute risk to algae	Ref 17: AARI	Algae LC/EC_{50}	Geometric mean of available data	AARI = sum for all active ingredients of (PEC/TOX) * area weighted average PEC = PECditch = mean dosage (kg/ha) * 0.4 * mean fraction drift; based on a model calculation TOX is acute toxicity to aquatic organisms Area-weighted average = kg a.i. sold / mean dosage

c. ALGAE

Index name	System ID	Input toxicity variables	Variable manipulation	Index manipulations
Risk to aquatic organisms (A)	Ref 22: Norwegian Indicator	Algae/water plant LC/EC_{50}	Not reported	$TER = LC/EC_{50} / PEC$ PEC is based on spray drift and surface runoff For studies with algae or water plants: Score 0: TER > 10 Score 1: TER = 10– 10 Score 2: TER = 0.1 – 1 Score 3: TER = 0.01 – 0.1 Score 4: TER < 0.01
Acute risk to water organism	Ref 23: POCER	Algae NOEC	Chose the lowest of these: LC_{50} for fish/100 EC_{50} for Daphnia/100 NOEC for algae/10	PEC/chosen toxicity PEC based on application rate, drift, width and depth of ditch
Short-term risk to algae	Ref 24: PRISW-1	Algae EC_{50}	Not reported	$TER_{algae} = LC_{50} / PEC_{short\text{-}term}$ PEC based on drift and runoff Result is scored from 0-8 PRISW-1 = [3 * Score(Algae)] + [4 * Score(Daphnia)] + [5.5 * Score(Fish)] For ERIP, all compartments are added but scoring and weights in each compartments change

c. ALGAE

Index name	System ID	Input toxicity variables	Variable manipulation	Index manipulations
Long-term risk to algae	Ref 24: PRISW-2	Algae NOEL	Not reported	TER_{algae} = NOEL/Theoretical Exposure in Water TEW based on fugacity model results, application rate, and persistence Result is scored from 0-8 PRISW-2 = [2 * Score(Algae) + 3 * Score(Daphnia) + 3 * Score(Fish)] * B * S Where B is the bioaccumulation potential (K_{ow}) S is the % distribution of substance in sediment (fugacity level 1) For ERIP, all compartments are added but scoring and weights in each compartments change
Aquatic toxicity	Ref 27: APPLES	Preferential use of Selenastrum capricornum EC_{50}	Most sensitive amongst fish, invertebrate or algae species	Score 1: $EC_{50}/LC_{50} > 100$ Score 4: EC_{50}/LC_{50} = 10 – 100 Score 7: EC_{50}/LC_{50} = 1 – 10 Score 10: EC_{50}/LC_{50} = 0.1 – 1 Score 13: EC_{50}/LC_{50} = 0.01 – 0.1 Score 16: $EC_{50}/LC_{50} < 0.01$
Toxicity score	Ref 28: WWF	LC_{50}	Not reported	Score based on Kovach's EIQ

d. MACROPHYTES

Index name	System ID	Input toxicity variables	Variable manipulation	Index manipulations
Risk to Lemna	Ref 12: p-EMA	Algae NOEL	(1) Lemna minor (2) Lemna gibba	TER = acute toxicity/PEC PEC based on Application rate; Soil type; Soil organic matter; Crop cover; Method of application

Figure 1: Basic structure of effect indices for mammals included in the reviewed systems (system ID); based on acute toxicity test results.

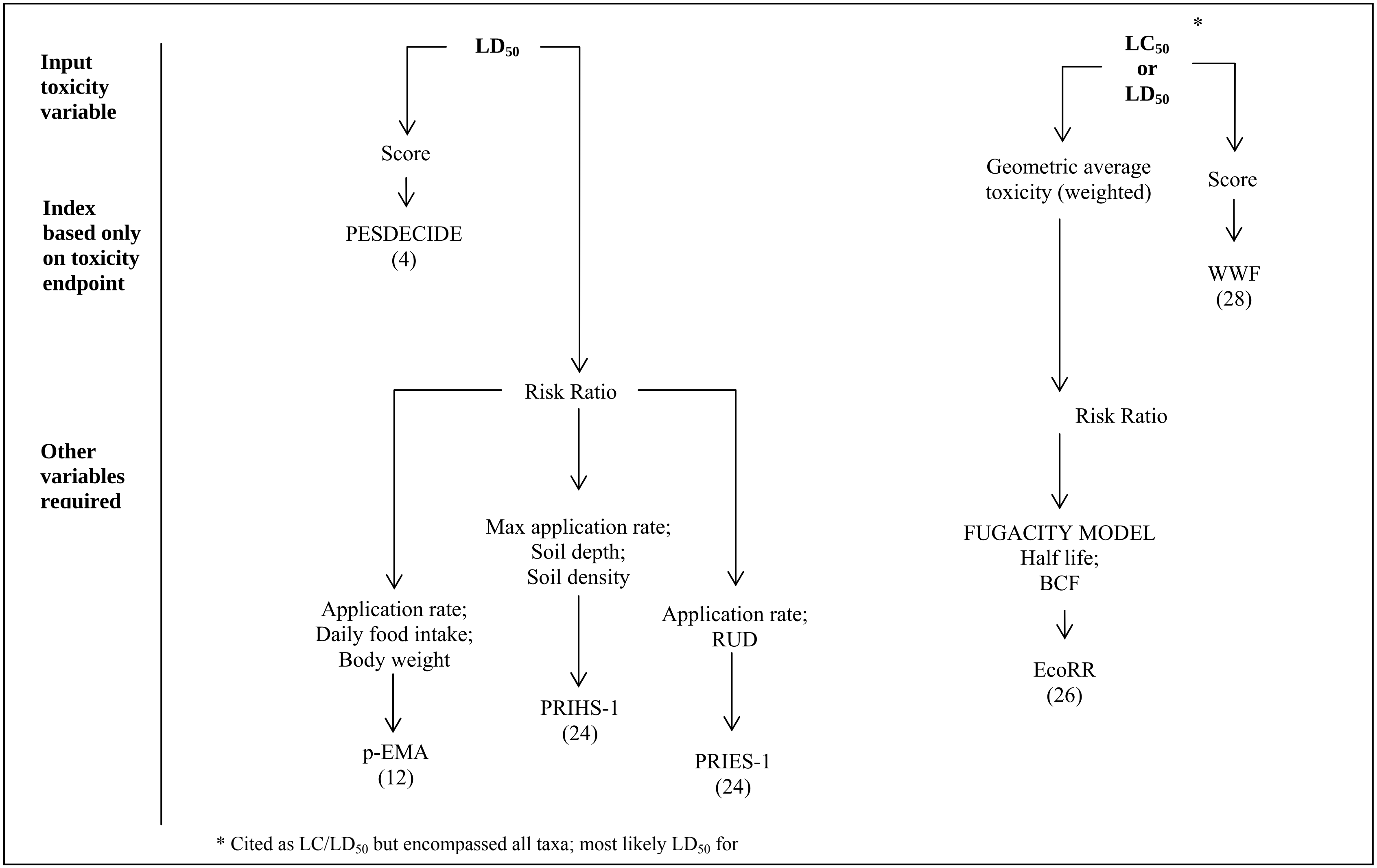


Figure 2: Basic structure of effect indices for mammals included in the reviewed systems (system ID); based on long-term toxicity test results.

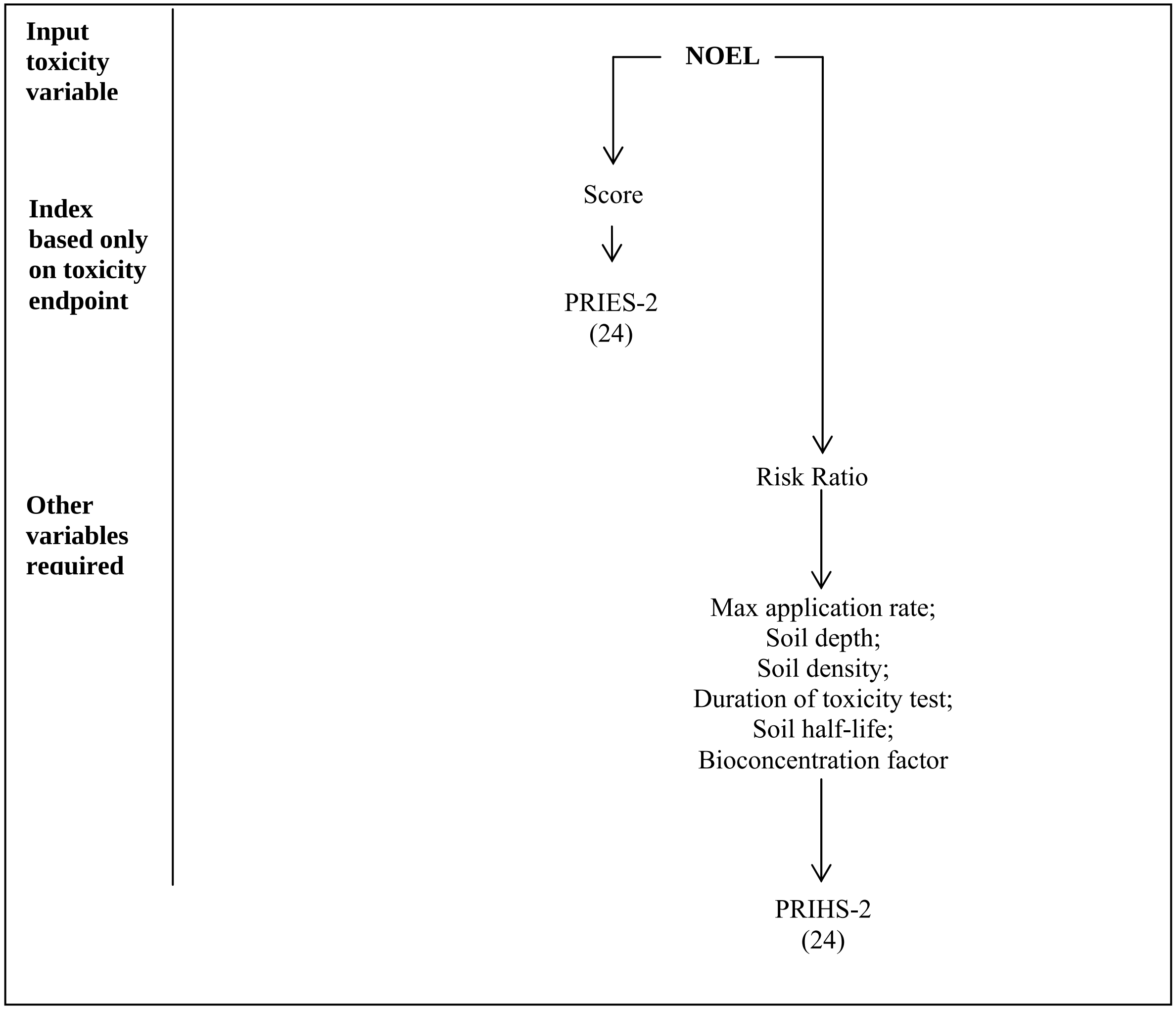


Figure 3: Basic structure of effect indices for birds included in the reviewed systems (system ID); based on short-term / acute toxicity

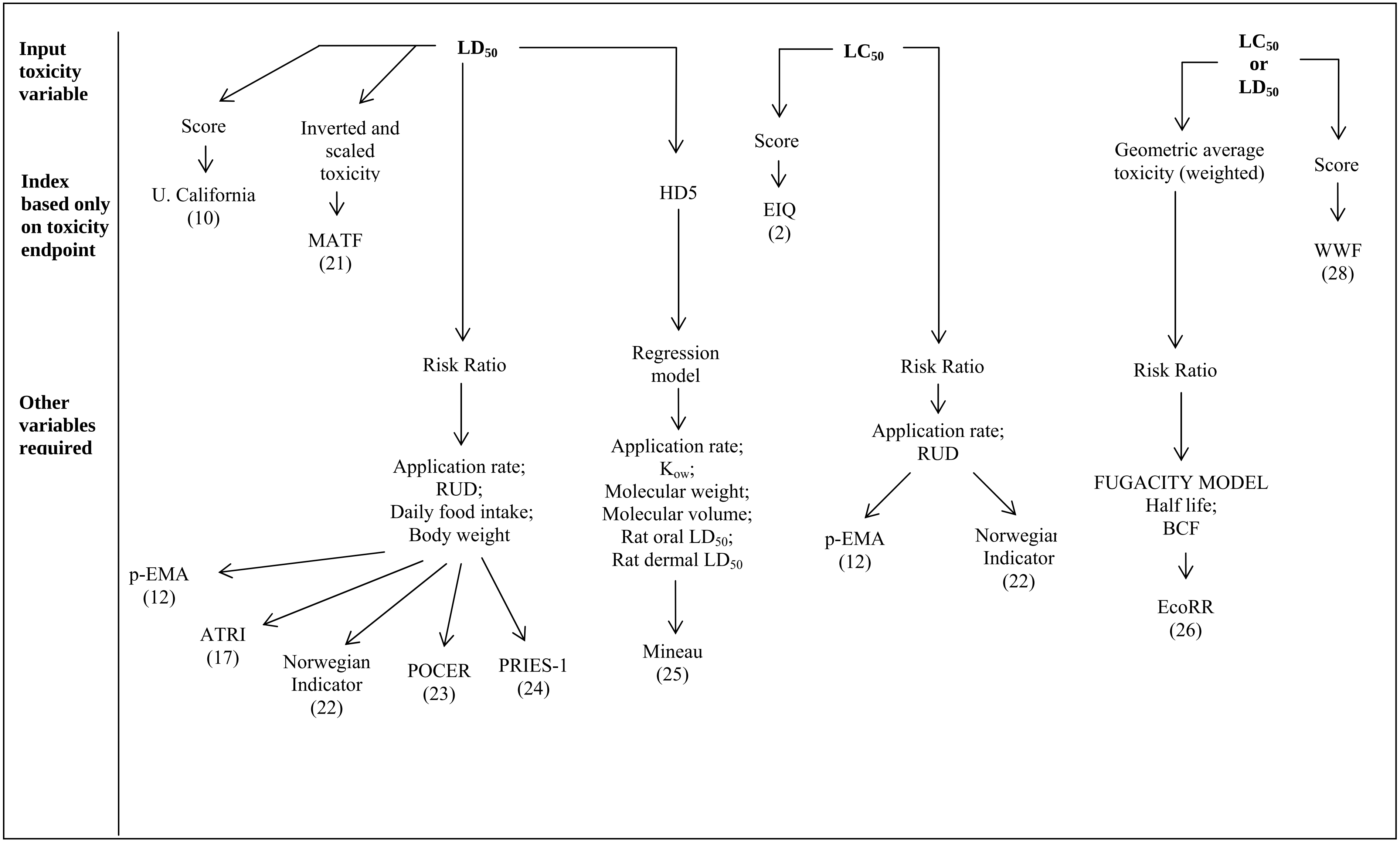


Figure 4: Basic structure of effect indices for bees included in the reviewed systems (system ID); based on acute toxicity test results.

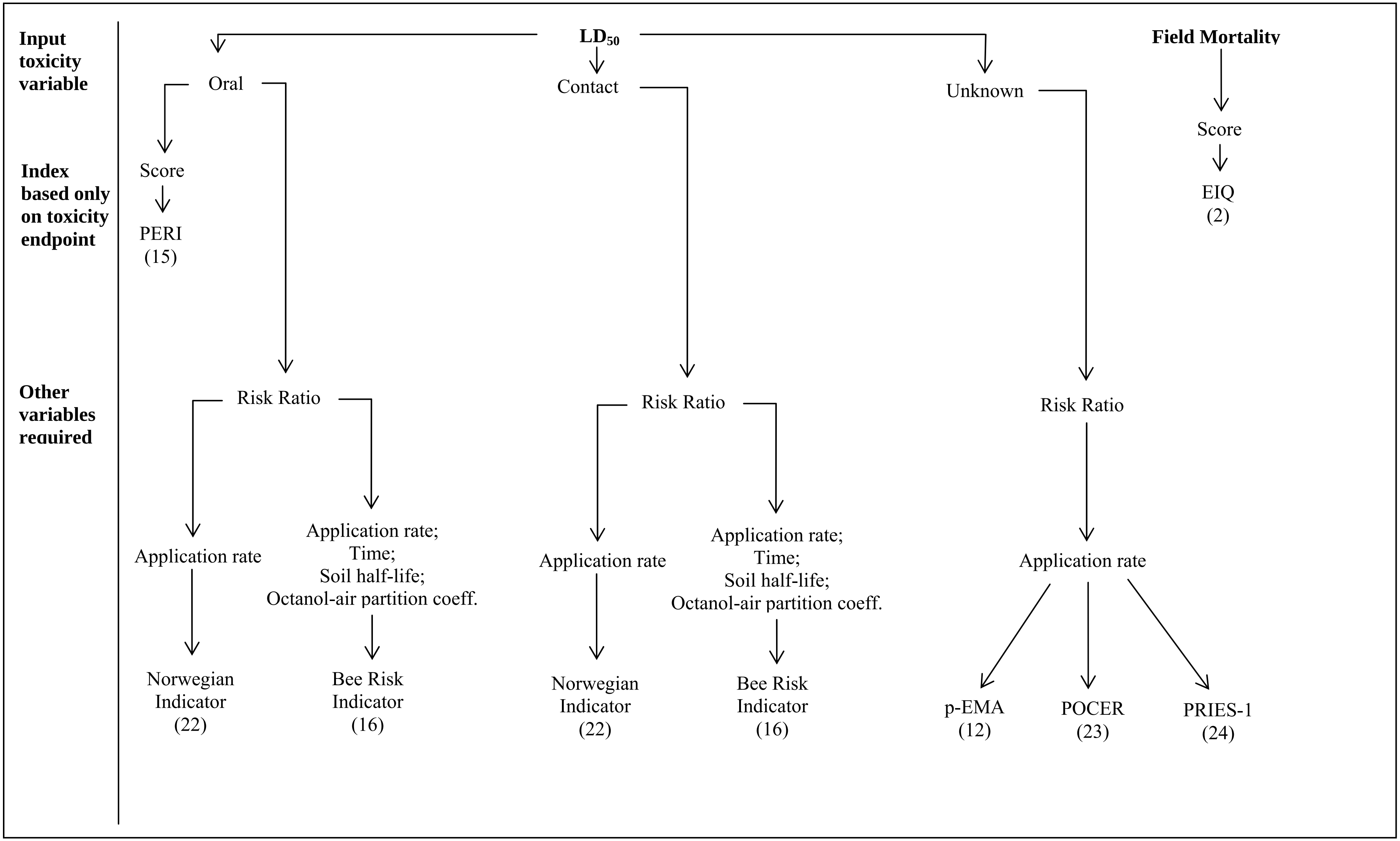


Figure 5: Basic structure of effect indices for birds, bees and beneficial arthropods included in the reviewed systems (system id) based on long-term toxicity test results.

Input toxicity variable	**NOEL** ↓ Score ↓
Index based only on toxicity endpoint	PRIES-2 (24)
Other variables required	

Figure 6: Basic structure of effect indices for earthworm included in the reviewed systems; based on acute toxicity test results.

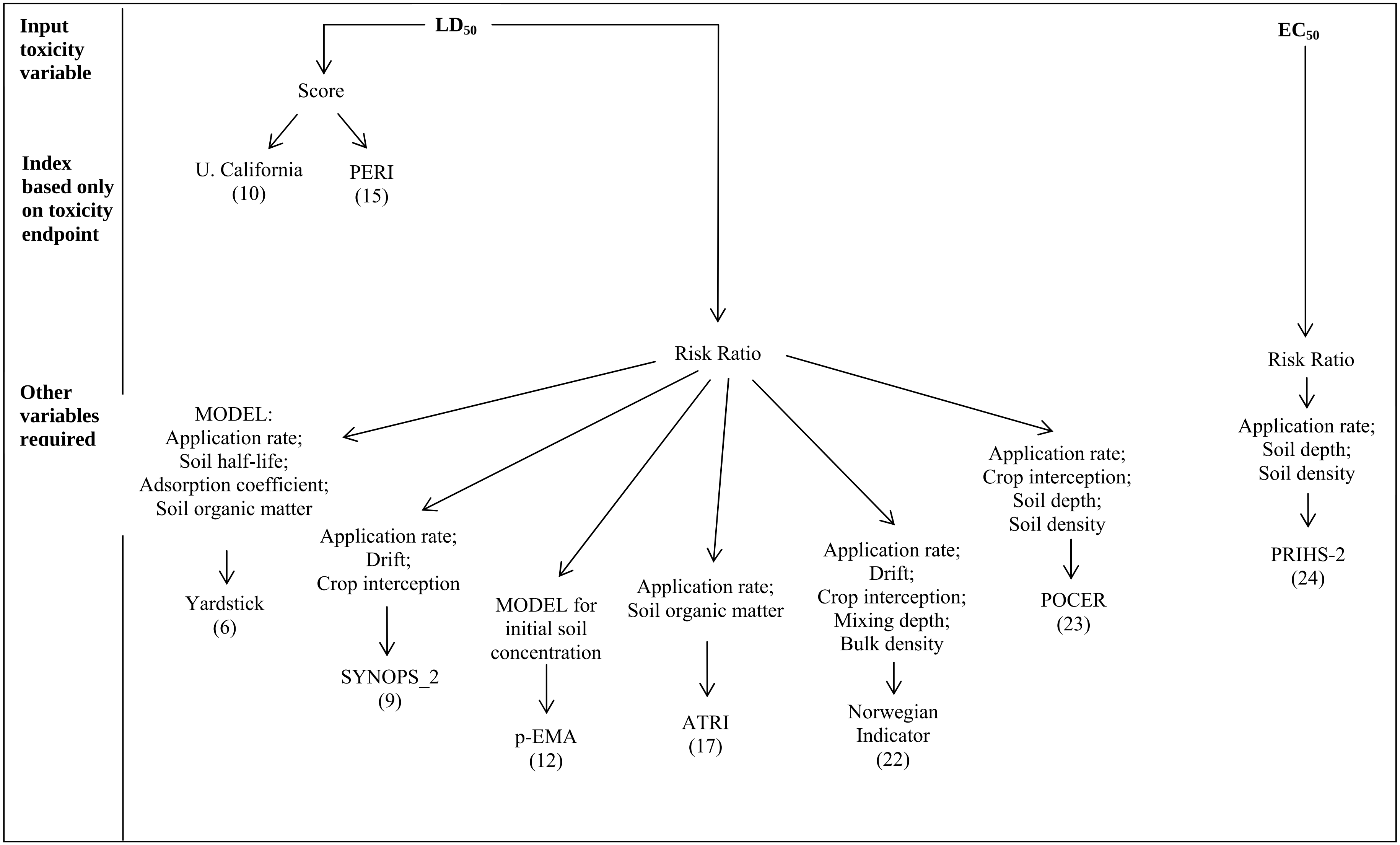


Figure 7: Basic structure of effect indices for earthworms included in the reviewed systems; (system ID); based on long-term toxicity test results.

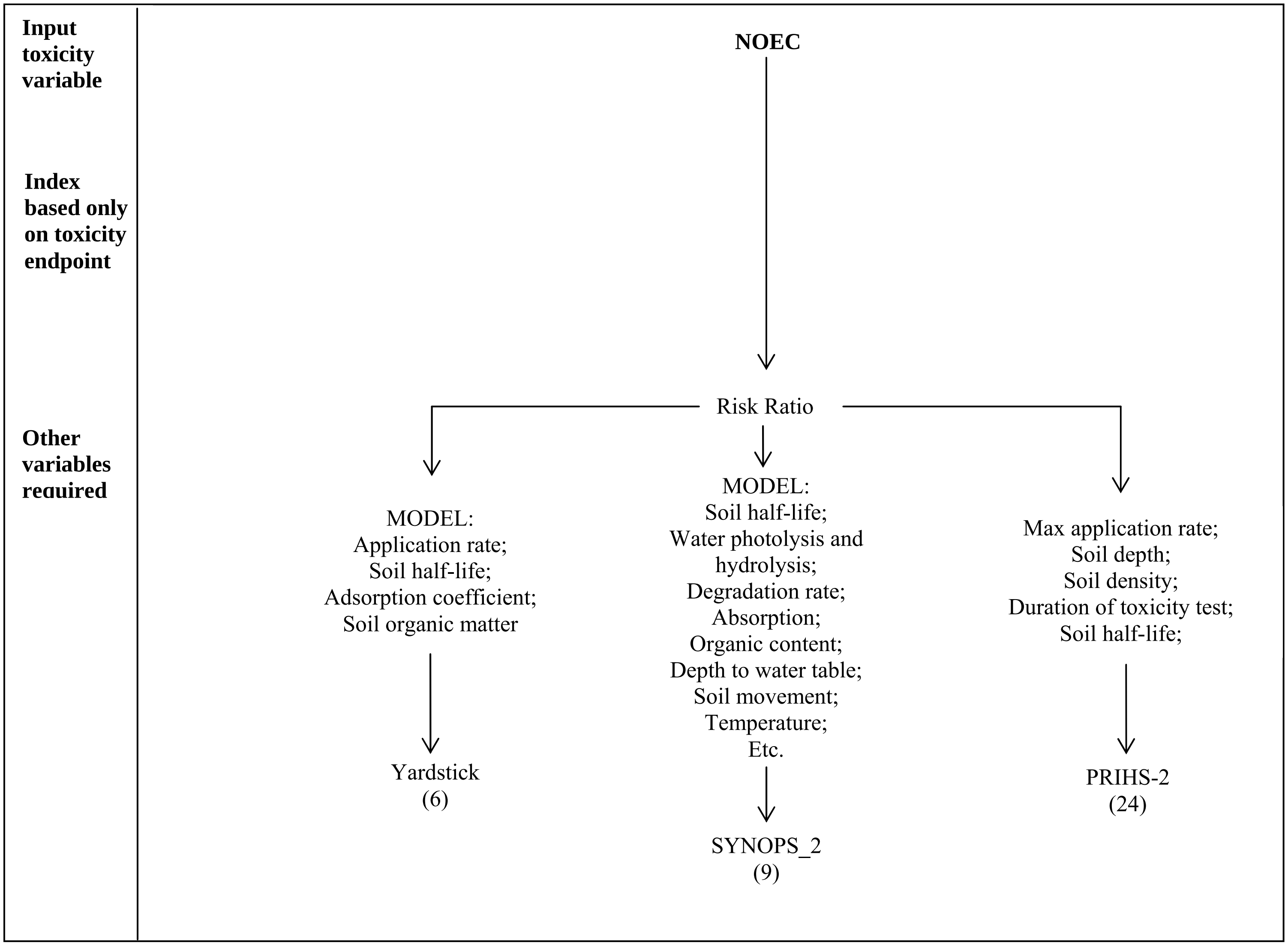


Figure 8: Basic structure of effect indices for earthworms included in the reviewed systems (system ID); based on long-term toxicity test results.

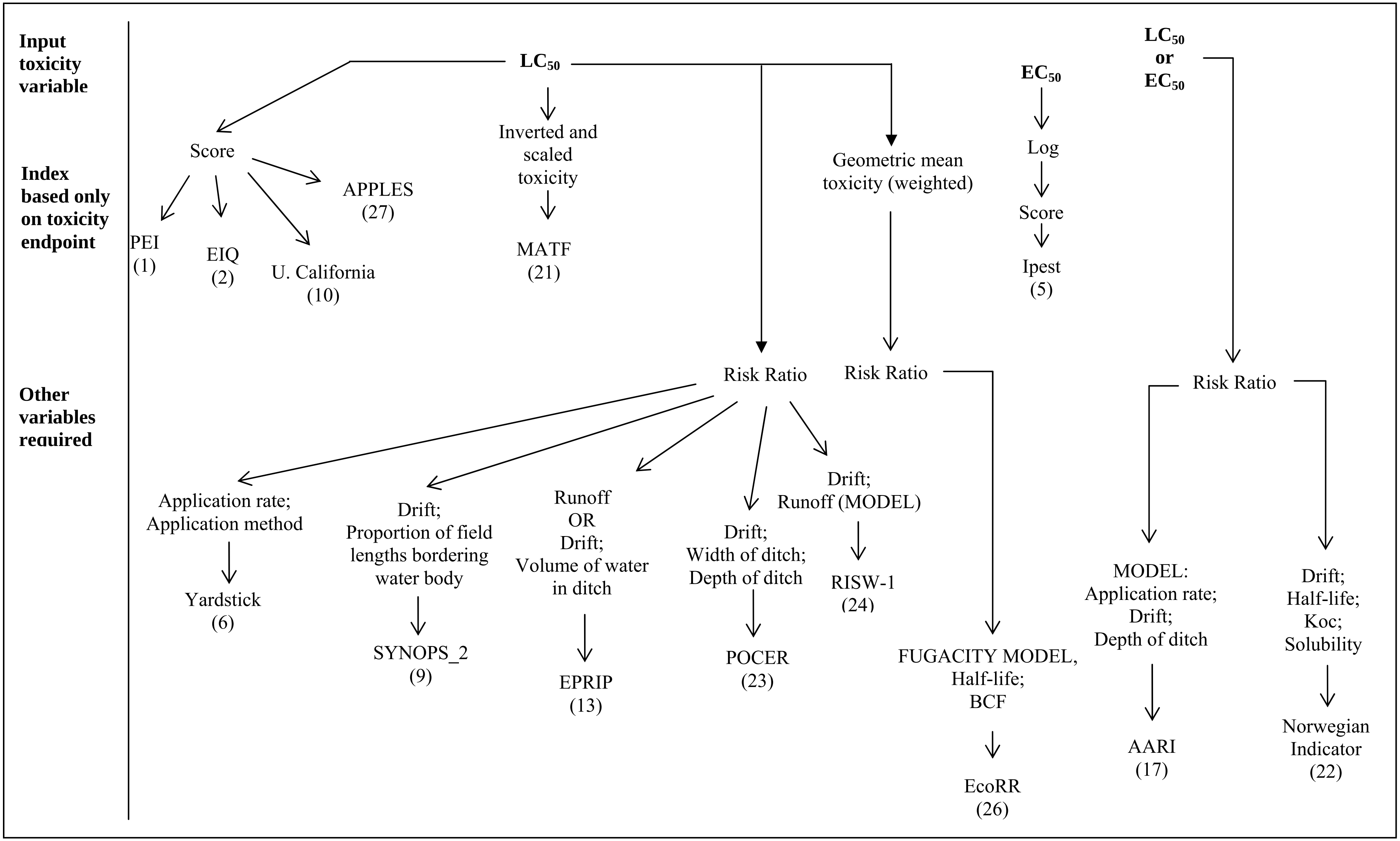


Figure 9: Basic structure of effect indices for fish included in the reviewed systems (system ID); based on long-term toxicity test results.

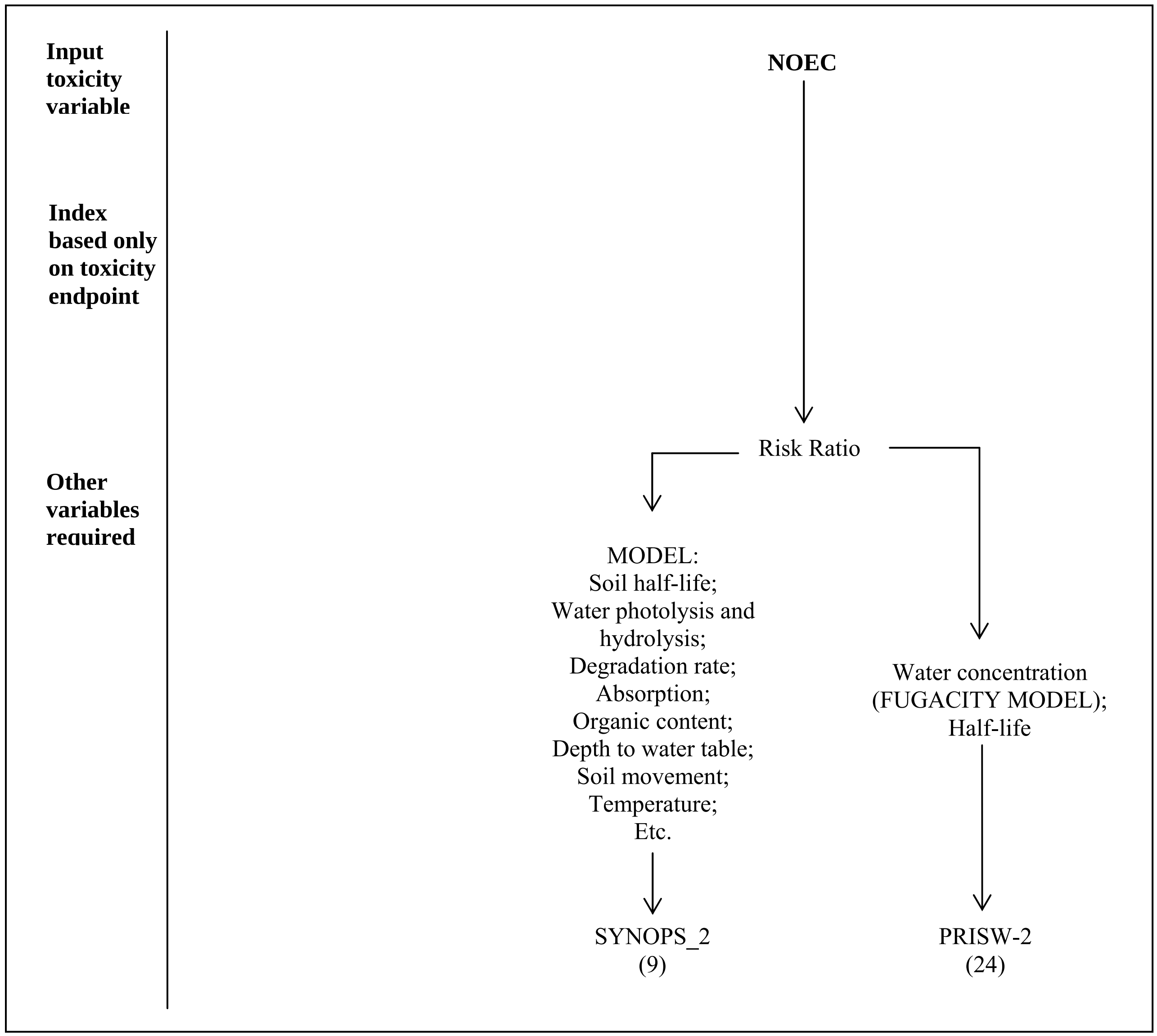


Figure 10: Basic structure of effect indices for aquatic invertebrates included in the reviewed systems (system ID); based on acute toxicity test results.

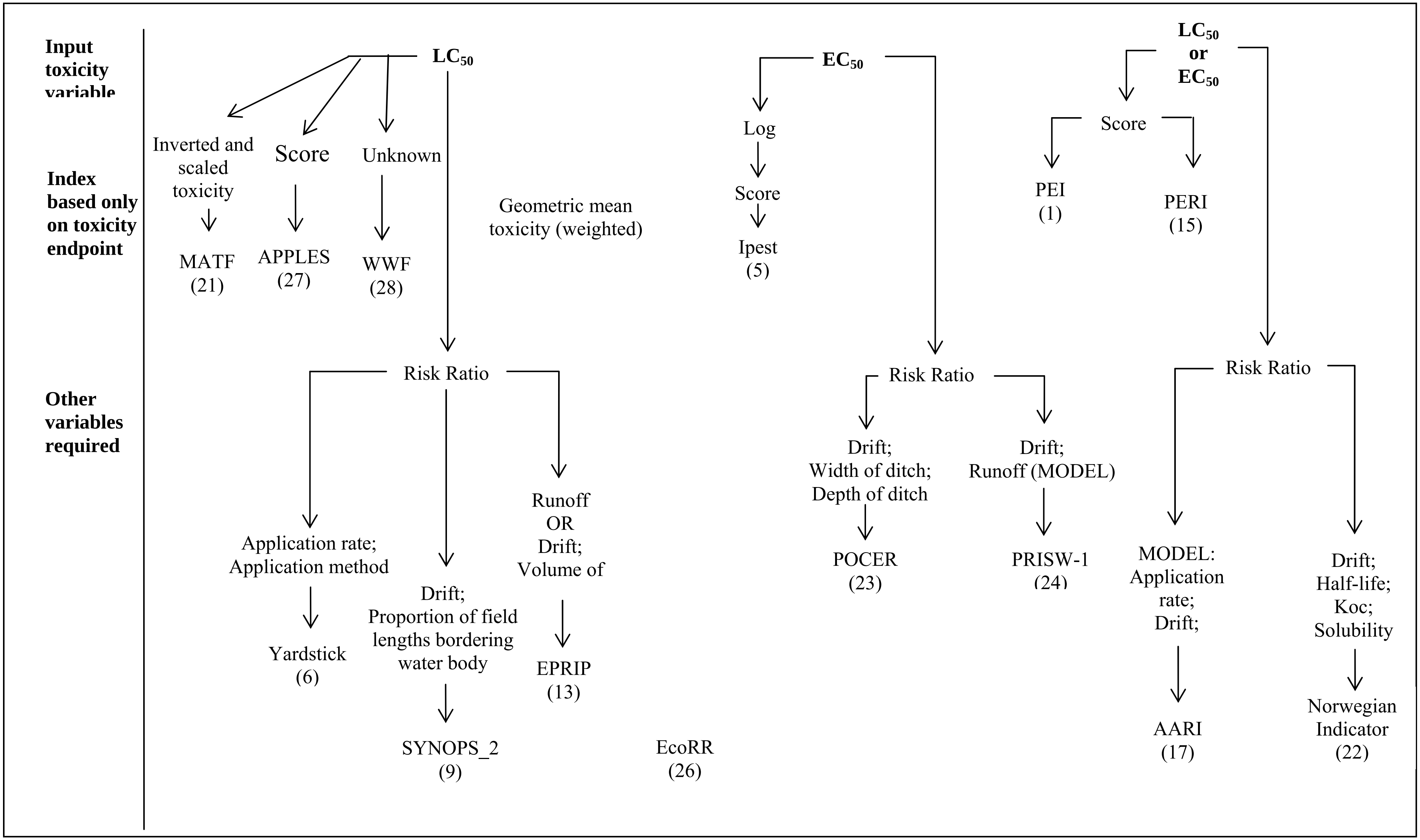


Figure 11: Basic structure of effect indices for aquatic invertebrates included in the reviewed systems (system ID); based on acute toxicity test results

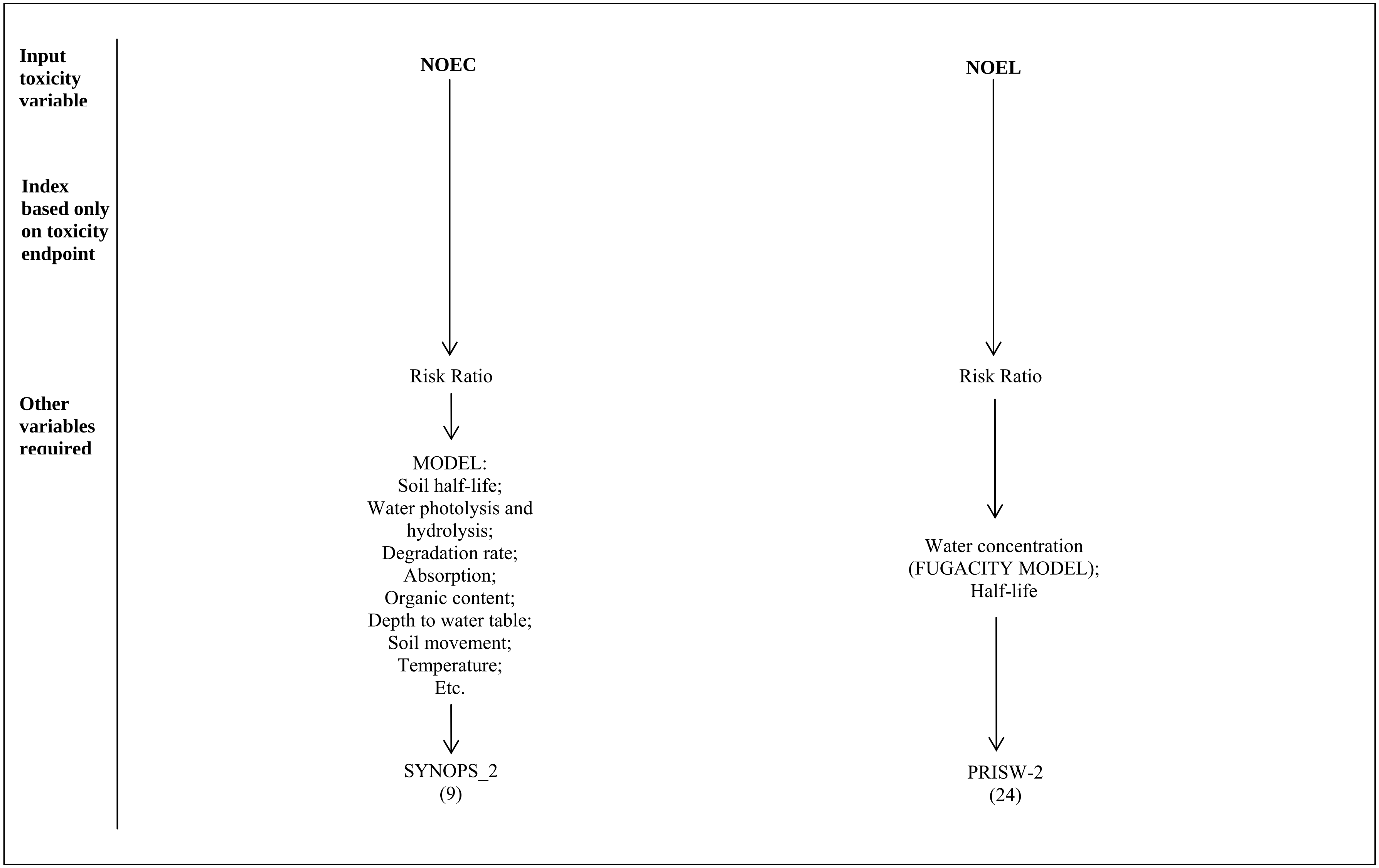


Figure 12: Basic structure of effect indices for algae included in the reviewed systems (system ID); based on acute toxicity test results.

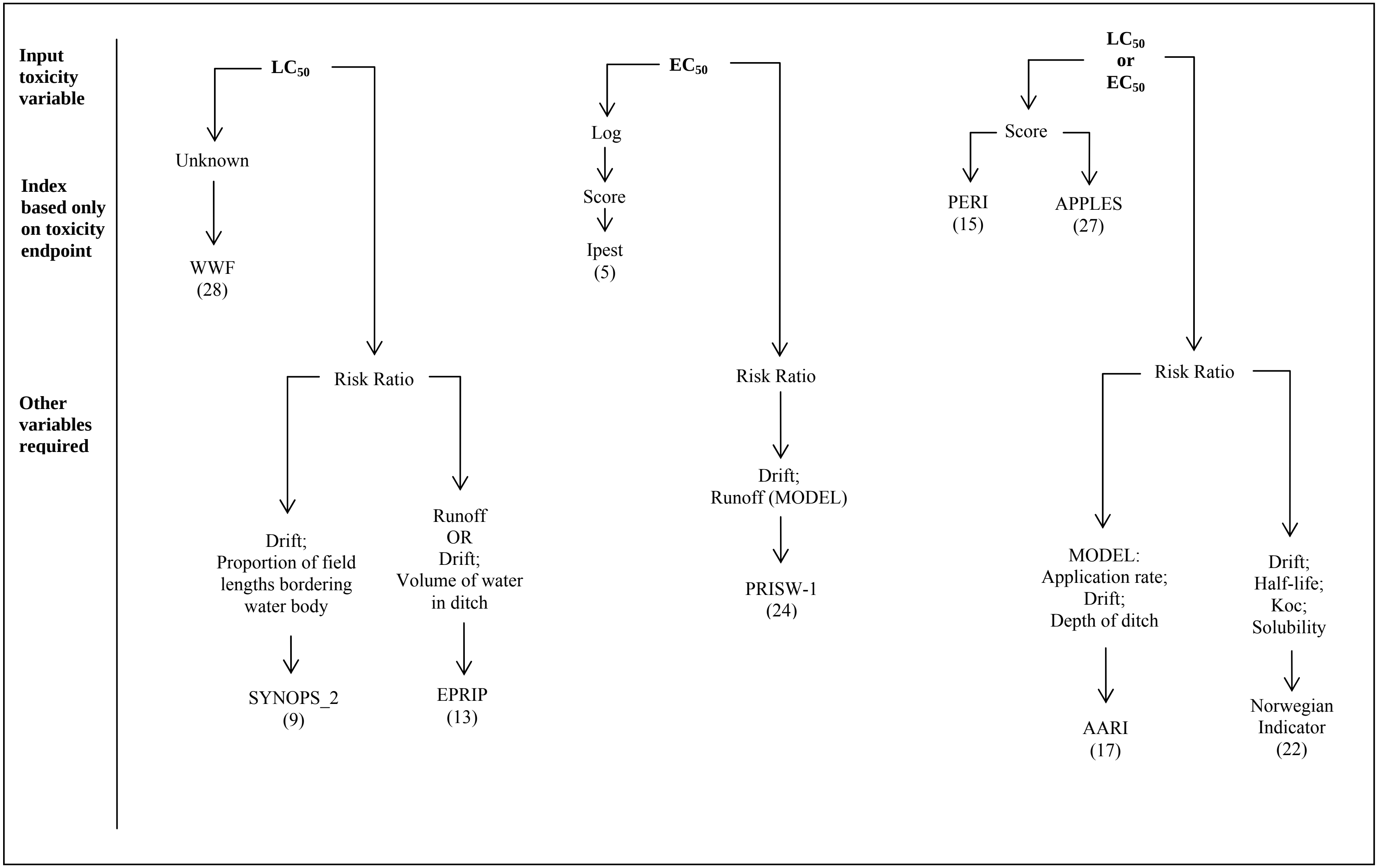
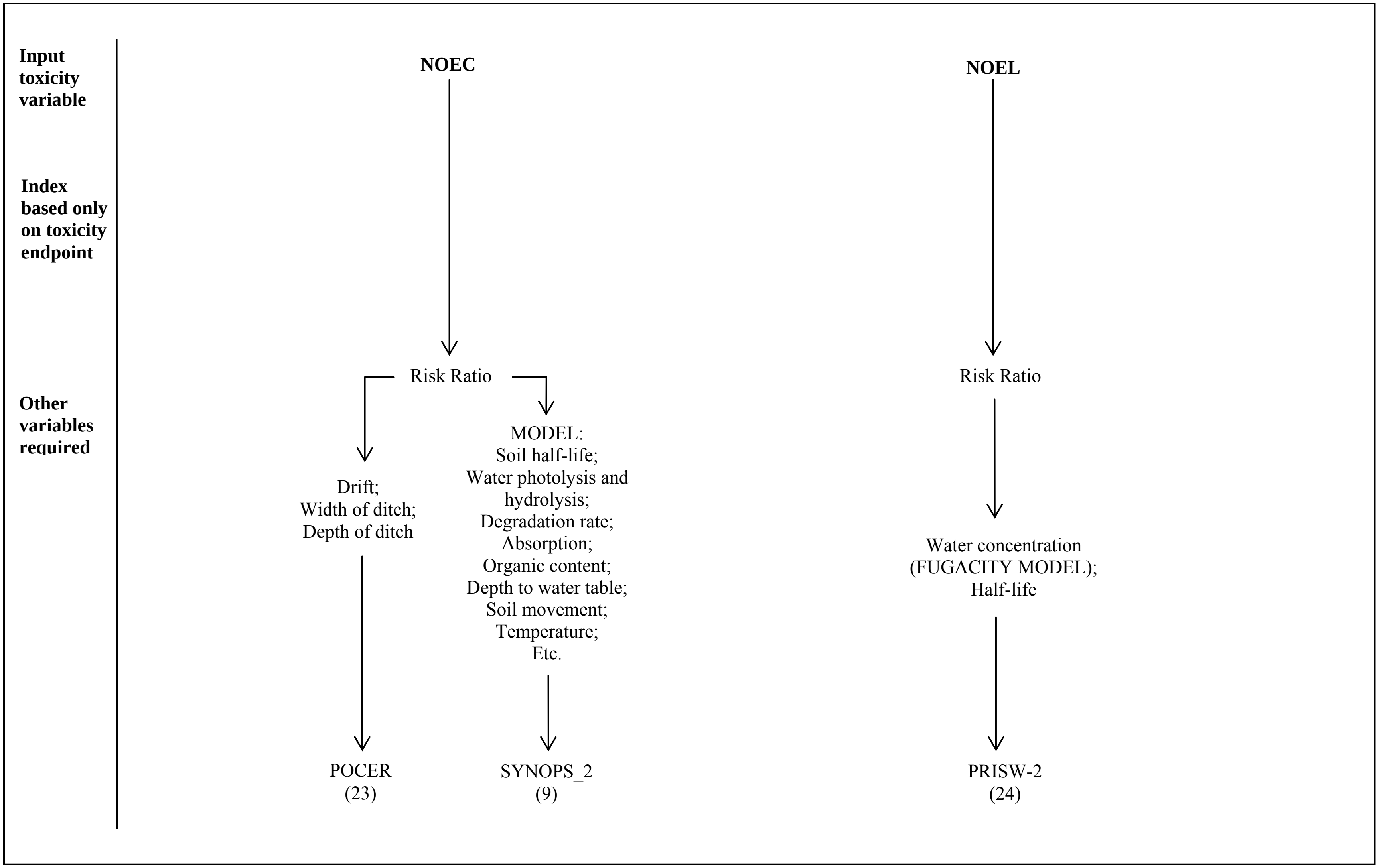


Figure 13: Basic structure of effect indices for algae included in the reviewed systems (system ID); based on long-term toxicity test results.



REFERENCES

Aldenberg, T., and W. Slob (1993). Confidence limits for hazardous concentrations based on logistically distributed NOEC toxicity data. Ecotoxicology and Environmental Safety 25:48-63.

Aldridge, C.A., and Hart A.D.M. (1993) Validation of the EPPO/CoE risk assessment scheme for honeybees. IN: Proceedings of the Fifth International Symposium on the Hazards of Pesticides to Bees, October 26-28 Plant Prot. Service, Wageningen, 37-41.

Baril, A., M. Whiteside, and C. Boutin. Analysis of a database of pesticide residues on plants for wildlife risk assessments. Environmental Toxicology and Chemistry In press.

Barnthouse, L. W. (2004). Quantifying population recovery rates for ecological risk assessment. Environmental Toxicology and Chemistry 23(2):500-508.

Benbrook, C., E. Groth III, J. M. Halloran, M. K. Hansen, and S. Marquardt (1996). Pest Management at the Crosswords. Consumers Union, Yonkers, NY, USA, 272 pages.

Benbrook et al. (2002). A Methodology for Adjusting Pesticide Use Data for the Toxicity of Active Ingredients. Available on the Pest Management at the Crossroads website (http://www.pmac.net/tox_unit_methods-2.pdf)

Bennett, R. S., I.C. Dewhurst, A. Fairbrother, A.D.M. Hart, M.J. Hooper, A. Leopold, P. Mineau, S.R. Mortensen, R.F. Shore, and T.A. Springer. A new interpretation of avian and mammalian reproduction toxicity test data in ecological risk assessment. Ecotoxicology In Press

Boutin, C., N. Elmegaard, and C. Kjær (2004). Toxicity testing of fifteen non-crop plant species with six herbicides in a greenhouse experiment: implications for risk assessment. Ecotoxicology 13:349-369.

Boutin, C., and C.A. Rogers (2000). Pattern of sensitivity of plant species to various herbicides – an analysis with two databases. Ecotoxicology 9:255-271.

Brock, T.C.M., J. Lahr, and P.J. Van den Brink (2000a). Ecological risks of pesticides in freshwater ecosystems. Part 1: Herbicides. Alterra Report 88. Alterra Green World Research, Wageningen, Netherlands, 124 pp.

Brock, T.C.M., R.P.A. Van Wijngaarden, and G.J. Van Geest (2000b). Ecological risks of pesticides in freshwater ecosystems. Part 1: Insecticides. Alterra Report 89. Alterra Green World Research, Wageningen, Netherlands, 142 pp.

Brown, C.D., A. Hart, K. Lewis, and I.G. Dubus (2003). p-EMA: simulating the environmental fate of pesticides for a farm-level risk assessment system. Agronomie 23:67-74.

Campbell, P.J., K.C. Brown, E.G. Harrison, F. Bakker, K.L. Barrett, M.P. Candolfi, V. Cañez, A. Dinter, G. Lewis, M. Mead-Briggs, M. Miles, P. Neumann, K. Romijn, R. Schmuck, S.

Shires, A. Ufer, and A. Watersdorfer (2000). A hazard quotient approach for assessing the risk to non-target arthropods from plant protection products under 91/414/EEC: hazard quotient trigger value proposal and validation. Journal of Pest Science 73:117-124.

Candolfi, M.P., F. akker, V. Cañez, M. Miles, C. Neumann, E. Pilling, M. Primiani, K. Rominj, R. Schmuck, S. Storck-Weyhermüller, A. Ufer, and A. Waltersdorfer (2000). Sensitivity of non-target arthropods to plant protection products: could Typhlodromus pyri and Aphidus spp. be used as indicator species? Chemosphere 39: 1357-1370.

Caux P-Y., and P. Jiapizian (2004). Reducing pesticide environmental risks: the National Agri-Environmental Standards Initiative (NAESI). Presentation: Aquatic Toxicity Workshop Charlottetown, October 24 – 27, (2004).

Chapman, P.M., R.S. Caldwell, and P.F. Chapman (1996). A warning: NOECs are inappropriate for regulatory use. Environmental Toxicology and Chemistry 15(2):77-79.

Cooney, J.D. (1995). Freshwater tests. In: G.M. Rand. (Ed.) Fundamentals of Aquatic Toxicology (Second Edition), Taylor and Francis, North Palm Beach, Florida, pp.71-102.

Crane, M., and M.C. Newman (2000). What level of effect is a no observed effect? Environmental Toxicology and Chemistry 19(2):516-519.

Duboudin, C.D., P. Ciffroy, and H. Magaud. (2004). Effects of data manipulation and statistical methods on species sensitivity distributions. Environm. Toxicol. Chem. 23(2):489-499.

Dunn, A.M. (2004). A relative risk ranking of pesticides used in Prince Edward Island. Environmental Protection Branch Surveillance Report EPS-5-AR-04-03, Environment Canada, Atlantic Region, Darmouth, Nova Scotia, 41pp.

European Commission (2002a). Guidance Document on Aquatic Ecotoxicology in the context of the Directive 91/414/EEC

European Commission (2002b). DRAFT. Guidance Document on Risk Assessment for Birds and Mammals Under Council Directive 91/414/EEC.

European Union (1997). Proposal for a Council Directive establishing Annex VI to Directive 91/414/EEC concerning the placing of plant protection products on the market, Official Journal of the European Communities, C 240:1-23

Finizio, A., M. Calliera, and M. Vighi (2001). Rating systems for pesticide risk classification on different ecosystems. Ecotoxicology and Environmental Safety 49:262-274.

Flari, V., J. Crocker, A. Hart, R. Ashauer, C. Brown, and A. Fliessbach. Inventory of existing pesticide risk indicators that include terrestrial risk indices. Harmonised environmental Indicators for pesticide Risk (HAIR), European Union, 55pp. Unpublished report.

Gutsche, V., and D. Rossberg (1997). SYNOPS 1.1: a model to assess and to compare the environmental risk potential of active ingredients in plant protection products. Agriculture Ecosystem and Environment 64:181–188.

Gutsche, V., and D. Rossberg (1998). SYNOPS_2 In: Integrated Pest Management Measurement Systems Workshop. Day, E. (Ed). Workshop held June 12-13 1998, Chicago, Illinois, pp. 99-113.

Hart A., C. D. Brown, K.A. Lewis, and J. Tzivilakis (2003). p-EMA (II): evaluating ecological risks of pesticides for a farm-level assessment system. Agronomie 23:75-84

Hose, G.C., and P.J. Van den Brink (2004). Confirming the species-sensitivity distribution concept for endosulfan using laboratory, mesocosm, and field data. Archives of Environmental Contamination and Toxicology 47:511-520.

Levitan, L. (1997). An overview of pesticide impact assessment systems (a.k.a. 'pesticide risk indicators') based on indexing or ranking pesticides by environmental impact. Background paper presented at the Organisation of Economic Cooperation and Development (OECD) workshop on pesticide risk indicators, 21-23 April 1997, Copenhagen, Denmark, 89pp.

Levitan, L. (2000). 'How to' and 'why': assessing the enviro-social impacts of pesticides. Crop Protection 19:629-636.

Lewis, M.A. (1995). Algae and vascular plant tests. In: G.M. Rand. (Ed.) Fundamentals of Aquatic Toxicology (Second Edition), Taylor and Francis, North Palm Beach, Florida, pp. 135-169.

Luttik, R., P. Mineau, and W. Roelofs. A review of interspecies toxicity extrapolation in birds and mammals and a proposal for long-term toxicity data. Ecotoxicology In Press

Mineau, P., B. Jobin, and A. Baril (1994). A critique of the avian 5-day dietary test (LC50) as the basis of avian risk assessments. Canadian Wildlife Service Technical Report Series 215. Canadian Wildlife Service, Environment Canada, Ottawa, ON, 23pp.

Mineau, P., B.T. Collins, and A. Baril (1996). On the use of scaling factors to improve interspecies extrapolation of acute toxicity in birds. Reviews of Environmental Contamination and Toxicology 170:13-74.

Mineau, P., and J. Duffe (2001). Birds and Insecticide Toxicoses – A Preliminary Risk Assessment for Israel. In Shlosberg, A. and O. Bahat. [eds.] The risks of toxicoses from pesticides and pollutants in raptors and wildlife in Israel. Published by the Society for the Protection of Nature in Israel, Tel Aviv, pp. 30-59.

Mineau, P. (2002). Estimating the probability of bird mortality from pesticide sprays on the basis of the field study record. Environmental Toxicology and Chemistry 21:1497-1506.

Mineau, P. A review and analysis of study endpoints relevant to the assessment of 'long term' pesticide toxicity in avian and mammalian wildlife. Ecotoxicology In Press

Mitchell, R.R., C.L. Summer, S.A. Blonde, D.M. Bush, K. Hurlburt, E.M. Snyder, and J.P. Giesy (2002). SCRAM: A Scoring and ranking system for persistent, bioaccumulative, and toxic substances for the North American Great Lakes – resulting chemical scores and ranking. Human and Ecological Risk Assessment 8:537-557.

Norwegian Agricultural Inspection Services (2000). Pesticide risk indicators for health and environment – Norway.

OECD (2000a). Report of the OECD Pesticide Aquatic Risk Indicators Expert Group. Available at http://www.oecd.org/document/45/0,2340,en_2649_34383_32265069_1_1_1_1,00.html

OECD (2000b). Survey of National Pesticide Risk Indicators. Available at http://www.oecd.org/document/45/0,2340,en_2649_34383_32265069_1_1_1_1,00.html

OECD (2004). DRAFT. Summary report of the OECD project on pesticide terrestrial indicators.

Reus, J.A.W.A, and P.C. Leendertse (2000). The environmental yardstick for pesticides: a practical indicator used in the Netherlands. Crop Protection 19:637-641.

Reus, J., P. Leendertse, C. Bockstaller, I. Fomsgaard, V. Gutsche, K. Lewis, C. Nilsson, L. Pussemier, M. Trevistan, H. van der Werf, F. Alfarroba, S. Blümel, J. Isart, D. McGrath, and T. Seppälä (1999). Comparing environmental risk indicators for pesticides. Results of the European CAPER Project. Centre for Agriculture and Environment (CML), The Netherlands, 184pp.

Reus, J., P. Leendertse, C. Bockstaller, I. Fomsgaard, V. Gutsche, K. Lewis, C. Nilsson, L. Pussemier, M. Trevistan, H. van der Werf, F. Alfarroba, S. Blümel, J. Isart, D. McGrath, and T. Seppälä (2002). Comparison and evaluation of eight pesticide environmental risk indicators developed in Europe and recommendations for future use. Agriculture, Ecosystems and Environment 90:177-187.

Sample, B.E., and C.A. Arenal (1999). Allometric models for interspecies extrapolation of wildlife toxicity data. Bulletin of Environmental Contamination and Toxicology 62:653-663.

Sánchez-Bayo, F., S. Baskaran, and I.R. Kennedy (2002). Ecological relative risk (EcoRR) : another approach for risk assessment to pesticides in agriculture. Agriculture, Ecosystems and Environment 91:37-57.

Sheehan, P., A. Baril, P. Mineau, and D. Paustenbach. (1995). Predicting the effects of insecticides on aquatic systems and the waterfowl that use them. In: G.M. Rand. (Ed.) Fundamentals of Aquatic Toxicology (Second Edition), Taylor and Francis, North Palm Beach, Florida, pp. 827-857.

Solomon, K.R., D.B .Baker, R.P. Richards, D.R. Dixon, D.R. Klaine, T.W. La Point, R.J. Kendall, C.P. Weisskopf, J.M. Giddings, J. P. Giesy, L.W. Hall Jr., and W.M. Williams (1996). Ecological risk assessment of atrazine in North American surface waters. Environmental Toxicology and Chemistry 15:31-76.

Stephan C.E., D.I. Mount, D.J. Hansen, J.H. Gentile, G.A. Chapman, and W.A. Brungs (1985). Guidelines for deriving numerical national water quality criteria for the protection of aquatic organisms and their uses. U.S. Environmental Protection Agency. PB85-227049, Springfield, VA. 98pp.

Sørensen, P.B., B.B. Morgensen, S. Gyldenkærne, and A.G. Rasmussen (1998). Pesticide leaching assessment method for ranking both single substances and scenarios of multiple substance use. Chemosphere 36:2251-2276.

Suter, G.W. (2002). North American history of species sensitivity distributions. In: Posthuma, L., Traas, T.P., Suter, G.W. II (eds) The use of species sensitivity distributions in ecotoxicology. Lewis, Boca Raton, Florida, pp. 11-17.

Swanson, M.B., G.A. Davis, G.A. Kincaid, L.E. Schultz, T.W. Bartmess, J.E. Jones, and E.L. George (1997). A screening method for ranking and scoring chemicals by potential human health and environmental impacts. Environmental Toxicology and Chemistry 16:372-383.

Teed, R.S. (2004). DRAFT. APPLES, a pesticide priority list: evaluation scheme for the CCME Water Quality Task group. Draft report prepared for the National Guidelines and Standards Office of Environment Canada and the Canadian Council of Ministers of the Environment and the National Guidelines and Standards Office.

Tomlin, C.D.S. (2000). The pesticide manual, 12th edition. The British Crop Protection Council, Binfield, Berks, UK, 1250pp.

Van Bol, V., P. Debongnie, L Pussemier, H. Maraite, and W. Steurbaut. (2002). Study and Analysis of Existing Pesticide Risk Indicators. Tervuren: Veterinary and Agrochemical Research Center (VAR). Unpublished report.

Van Gestel, C.A.M. (1992). Validation of earthworm toxicity tests by comparison with field studies: a review of benomyl, carbendazim, and carbaryl. Ecotoxicology and Environmental Safety 23:221-236.

Van den Brink, P.J., L. Posthuma, and T.C.M.Brock (2002). The value of the species sensitivity distribution concept for predicting field effects: (Non-) confirmation of the concept using semifield experiments. In: Posthuma, L., Traas, T.P., Suter, G.W. II (eds) The use of species sensitivity distributions in ecotoxicology. Lewis, Boca Raton, Florida, pp.155-193.

Van der Werf, H.M.G., and C. Zimmer (1998). An indicator of pesticide environmental impact based on a fuzzy expert system. Chemosphere 36:2225-2249.

van Straalen, N.M., and C.A. Denneman (1989). Ecotoxicological evaluation of soil quality criteria. Ecotoxicology and Environmental Safety 18:241-251.

Vercruysse, F., and W. Steurbaut (2002). POCER, the pesticide occupational and environmental risk indicator. Crop Protection 21:307-315.

Villa. S., M. Vighi, A. Finizio, and G.B. Sereni. (2000). Risk assessment for honeybees from pesticide-exposed pollen. Ecotoxicology, 9:287-297.

Wogram, J., and M. Liess (2001). Rank ordering of macroinvertebrate species sensitivity to toxic compounds by comparison with that of Daphnia magna. Bulletin of Environmental Contamination and Toxicology 67:360-367.

World Wildlife Fund (2003). Recommendations for prioritizing risk reduction strategies based on toxicity loading by commodity, by region, and by pesticide: a discussion draft.

APPENDICES

APPENDIX A: Overview of models

Information presented here should not be considered as full model descriptions. Only the information relevant to the discussion presented in this report is detailed here. For full model descriptions, refer to the references cited in Table 1.

A RELATIVE RISK RANKING OF PESTICIDES USED IN PEI

Environment Canada, Atlantic Region
Dunn (2004)

General notes:

- To develop a priority listing of pesticides to direct and prioritize future pesticide risk assessments and management activities.
- Is based on the CHEMS risk ranking model (Swanson et al., 1997)

Model structure:

- Based on the principle of risk = hazard * exposure potential
- Both hazard and exposure are scores; each is in fact a composite index obtained from an a addition of scores (see below)

Elements of the model:

1. Hazard = Score(Human health hazard) + Score(Environmental hazard)
   a Human health hazard
   Adds scores (ranging from 1 to 5) for 4 variables
   Beyond the scope of NAESI – see Dunn 2004 for details
   b Environmental hazard
   Variables are
   - Fish 96-hour LC_{50}
   - Daphnia 48-hour LC_{50} or EC_{50} (immobilization)

   Scoring criteria for acute fish and Daphnia toxicity:

- Score 0: $\geq$ 1000 mg/L
- Score 5: < 1 mg/L
- Between cut-off values:
- Score = -1.67log (LC_{50} or EC_{50}) +5.0

Manipulations:

- Environmental hazard = sum of HV for fish and Daphnia

2. Exposure potential = Score(BCF) + Score(soil half-life)

a Bioconcentration

Score from 1 to 2.5

Scoring criteria:

- Score 1: log (BCF) $\leq$ 1.0
- Score 2.5: log (BCF) > 4.0
- Between cut-off values:
- Score = 0.5log (BCF) + 0.5

Missing data:

- If the bioconcentration factors were not available in the literature, they were estimated by a QSAR developed by Bintein et al. (1993, cited)

b Soil half-life

Score from 1 to 2.5

Scoring criteria:

- Score 1: $\leq$ 4 days
- Score 2.5: > 500 days
- Between cut-off values:
- Score = 0.311ln (Soil half-life) + 0.568

When test conditions reported, used soil half-life values for sandy loam for this type of soil is representative of those in P.E.I.

In the absence of soil half-life data, used the field dissipation half-life (time required for a substance to dissipate to half its original concentration under field conditions)

In the original CHEMS model, persistence (represented here by soil half-life) was represented by water half-life

$$\text{Water half-life} = \frac{1}{(1/\text{BOD half-life} + 1/\text{Hydrolysis half-life})}$$

The biological oxygen demand (BOD) half-life is the time required to biodegrade a chemical such that its BOD in water is reduced by half. But these values are not readily available in the literature. It was first proposed that BOD half-life be replaced by soil half-life in the water half-life equation. But an active ingredient with a quick hydrolysis rate could then appear to be less persistent than by considering soil-half-life alone. Also, since soil contains pore water, presumably hydrolysis is already accounted for in the soil half-life. Soil half-life alone was therefore considered.

3. Risk = Score(Hazard) * Score(Exposure Potential)

   Max score of 150

4. Release Weighting Factor (RWF)

   - weight applied to hazard scores
   - soil half-life and bioaccumulation parameters were not weighted as they already represent exposure potential
   - is media specific: level III fugacity model (Mackay 1991, cited)
   - release amount (in kg) is based on pesticide sales data in PEI
   - Score from 1 to 10

   Calculations:

   RWF_m = ln [release amount$(kg)_m$] + a

   Where a = 10 – ln [max release amount$(kg)_m$] and m = media of interest (water is of interest for environmental hazard)

   If release amount (kg) below $e^{(1-a)}$, then $RWF_m = 1$, (this cut-off value is used to avoid negative factors)

5. Weighted risk = same as above except that scores for hazard are multiplies with weighting factors

THE ENVIRONMENTAL IMPACT QUOTIENT (EIQ)

Kovach et al., 1992 as reviewed by Levitan (1997)

General notes:

- For IPM specialists to aid fruit and vegetable growers chose low impact options.
- Designed for growers in New York State

Model structure:

- Sum of the scores of all indices (scores may be affected by a weighting factor); therefore end result is a composite score

Elements of the model:

1. Risk to applicators
   Multiplies scores (1 or 3 or 5) for at least two variables
   Full description is beyond the scope of NAESI
   Max score of 125
2. Risk to pickers
   As above
   Max score of 25
3. Food residues
   As above
   Max score of 75
4. Leaching potential in ground water (classified it as a health issue)
   As above
   Max score of 5
5. Impact on aquatic vertebrates
   Variables are:
   - Fish Toxicity (96h LC50)
   - Surface loss potential

   Scoring criteria for fish toxicity:
   - Score 1: > 10 ppm
   - Score 3: 1 – 10 ppm
   - Score 5: < 1 ppm

Potential of surface runoff reaching fish is based on water half-life, solubility, adsorption coefficient, and soil properties. Scoring criteria are:

- Score 1: small
- Score 3: medium
- Score 5: large

Manipulations:

- Score(Fish toxicity) * Score(Surface runoff potential)

Max score of 25

6. Impact on birds

Variables are

- Bird Toxicity (LC50)
- Soil half-life
- Plant surface residue half-life

Scoring criteria for bird toxicity:

- Score 1: > 1000 ppm
- Score 3: 100 – 1000 ppm
- Score 5: 1 – 100 ppm

Scoring criteria for soil residue half-life:

- Score 1: < 30 days
- Score 3: 30 – 100 days
- Score 5: > 100 days

Scoring criteria for plant residue half-life:

- Score 1: 1-2 wks
- Score 3: 2-4 wks
- Score 5: > 4 wks

Manipulations:

- Score(Bird Toxicity) * [0.5 * Score(Soil half-life) + 0.5 * Score(Plant half-life)] * 3

Max score of 75

7. Impact on bees

Variables are

- Lethality to honey bees at field doses
- Plant surface residue half-life

Scoring criteria for bee lethality:

- Score 1: relatively non toxic
- Score 3: moderately toxic
- Score 5: highly toxic

Scoring criteria for plant surface residue half-life above, as for birds

Manipulations:

- Score(Lethality to honey bees) * Score(Plant half-life) * 3

Max score of 75

8. Impact on beneficial arthropods

Variables are

- Beneficial arthropod toxicity
- Plant surface residue half-life

Scoring criteria for beneficial arthropod toxicity:

- Score 1: low impact
- Score 3: moderate impact or post-emergent herbicides
- Score 5: severe impact

Scoring criteria for plant surface residue half-life as above

Manipulations:

- Score(Beneficial arthropod tox) * Score(Plant half-life) * 5

Max score of 125

Other notes:

The set weighting factors are not the only source of weighting; the number of variables for each index will affect the max score (thus weight) of an index in the final equation – even if weighting factor is 1, the max score can be higher than 5 if more than one variable for manipulations.

Data gaps for variables are filled using the average score for the pesticide class (insecticide, herbicide, fungicide).

The basic EIQ final score can be adjusted to situation-specific variables:

- EIQ Field Use Rating = EIQ * application rate * % a.i. in product
- EIQ are a.i. specific but the EIQ Field Use Rating is product specific – because concentration of active ingredient and recommended dosage are both specific to a particular trade product. However, the potency of the active ingredient may vary in different formulations due to adjuvants and 'inert ingredients'.

Cited reference:

Kovach, J., C. Petzoldt, J. Degni, and J. Tette (1992). A method to measure the environmental impact of pesticides. New York's Food and Life Sciences Bulletin 139:1-8

STEMILT GROWERS INTEGRATED FRUIT PRODUCTION RESPONSIBLE CHOICE POINT SUMMARY

Stemilt Growers, a fruit packing and marketing company in Washington State
As reviewed by Levitan (1997)

General notes:

- Is intended both as a guide to farmers toward IPM and for eco-labelling
- Is pest specific rather than specific to active ingredient only; therefore pesticide ratings can only be compared when used to combat the same pest.
- Stemilt Growers have derived point summaries for each pest their contract growers are likely to encounter; the assumption is pesticides are used at label rate – if less is used, a proportional number of points are assigned. Growers are supplied with a handbook.

Model structure:

Is a composite point system; sum of ratings for 8 indicators (which are affected by a weighting factor)

Elements of the model:

1. Efficacy

    Is based on a subjective comparison of available options

    Used as an indirect measure of dosage and needed number of applications

    Points from 1-4

    Weighting factor of 3
2. Toxicity to farm workers

    Points from 0-3

    Weighting factor of 1
3. Consumer exposure (i.e. the amount of time legally required between the last field application and harvest)

    Points is days/7

    Weighting factor of 2
4. Leaching potential

    Points from 0-3

    Weighting factor of 2

5. Soil sorption

    Points from 1-3

    Weighting factor of 1

6. Soil half life

    Points is days/20

    Weighting factor of 1

7. Acute impact on beneficials

    Points from 0-5

    Weighting factor of 1

    The variable used is described as “effect on beneficials”. Is a measure of acute toxicity to insects in the field at the time of application

8. Long-term impact on beneficials

    Points from 0-25

    Weighting factor of 1

    Based on “biological disruption”

    Model is for fruit production in the Pacific North-West – i.e. data underlying the ratings for efficacy and impact on beneficials are particular to growing conditions in that environment.

PESTDECIDE©

Australia
As reviewd by Levitan (1997)

General notes:

- To guide growers toward IPM and for eco labelling
- Is pest-specific
- Sensitive to formulations, not only active ingredient
- Scores presented to growers as tables in a manual; growers also record spray schedule

Model structure:

Sum of ratings assigned to 10 variables (which are affected by a weighting factor)

Elements of the model:

1. Activity

   Score will increase with the highest concentration recommended for field spray, on the premise that more active compounds will require lower dosages and are thus less likely to produce residues. A criteria matrix exists (Penrose et al, 1995b)

   Score from 1-5

   Weighting factor from 1-4

2. Site of application

   Score 1: when ground application

   Score 2: application on dormant, non-bearing or post-harvest trees

   Score 3: application on blossoms

   Score 4: application on fruit on trees (petal fall to harvest)

   Score 5: application on post-harvest fruit

   Weighting factor from 1-4

3. Timing of application

   Score 1: last application during dormancy or post-harvest trees

   Score 2: last application during first-half of growing season

   Score 3: last application during second half of growing season

   Score 4: last application at < 7 days before harvest

   Score 5: post-harvest fruit dip

Weighting factor from 1-4

4. Persistence (is essentially the preharvest interval)

    Score 1: there is no legally-mandated waiting period

    Score 2: 1-3 days

    Score 3: 4-14 days

    Score 4: 15-42 days

    Score 5: > 42 days

    Weighting factor from 1-4

5. Efficacy

    A subjective judgment of efficacy of treatment in comparison with general expectations for modern pesticides, across all products and targets

    More effective products are assigned lower scores; where info is lacking, score of 3.

6. Cost

    Score 1: < 1$ per 100L

    Score 2: 1.01$ - 2.00$ per 100L

    Score 3: 2.01$ - 3.00$ per 100L

    Score 4: 3.01$ - 4.00$ per 100L

    Score 5: > 4.00$ per 100L

    Weighting factor from 1-4

7. Environmental effect

    Based on the EIQ (developed by Kovach)

    Score 1: EIQ 0-25

    Score 2: EIQ 26-35

    Score 3: EIQ 36-45

    Score 4: EIQ 46-60

    Score 5: EIQ >60

    Weighting factor from 1-4

8. Mammalian toxicity: dermal LD_{50}

    Score 1: LD_{50} > 1000 mg/kg

    Score 2: LD_{50} = 501 – 1000 mg/kg

    Score 3: LD_{50} = 51 – 500 mg/kg

    Score 4: LD_{50} = 5 – 50 mg/kg

    Score 5: LD_{50} < 5 mg/kg

Weighting factor from 1-4

9. Compatibility with IPM

    The degree of disruption to biological control of other pests

    Is rated on a subjective scale with lower scores for least disruption

    Score from 1-5

    Weighting factor from 1-4

10. Availability of alternative pesticides

    Lower scores when fewer viable alternatives exist

    Score 5: > 4 alternative active ingredients are available

    Weighting factor from 1-4

IPEST: PESTICIDE ENVIRONMENTAL IMPACT INDICATOR BASED ON A FUZZY EXPERT SYSTEM

INRA, France
Van der Werf and Zimmer (1998)
Also reviewed by Levitan (1997) and CAPER

General notes:

- To help farmers choose the safest pesticide for particular field conditions
- Is based on the fuzzy expert system. For each variable, authors have defined membership criteria for 2 fuzzy subsets: F (favourable, no potential environmental impact) and U (unfavourable, max potential for environmental impact). Criteria (threshold values) are based on literature data, judgement of the authors, or on input from the end-users of the system. A value can fall within the range of safety (thus a member of subset F) or within the range defined for maximum potential for negative impact (subset U). Members of the F and U subsets are attributed a score of 0 and 1, respectively. When a value falls between the threshold values, it is said to have partial membership in both subsets and will have a score anywhere between 0 and 1, depending on its degree of membership to either subsets. Membership is assigned based on a sinusoidal function

Model structure:

The model is in a form of a decision tree

Elements of the model:

1. Presence of the pesticide in environment

   Based on the log (application rate active ingredient)

   F: < 1 (10 g a.i./ha)

   U: > 4 (10 000 g a.i./ha)

2. Risk of surface water contamination:

   a Runoff potential

   Based on users' assessment of field slope and distance to water edge

   F: no potential for runoff

   Membership in F and U: some potential for runoff

   U: major potential for runoff

b Drift potential

Based on user's assessment to distance to water edge and application technology

F: 0%

U: > 1%

c Position of application

F: in soil or on seed or on crop

Membership in F and U (if not in soil): (100 - % soil covered by crop)/100

U: on soil surface

d Persistence

Based on the Soil degradation rate (DT_{50})

F: < 1 day

U: > 30 days

e Toxicity

Based on toxicity to most sensitive aquatic organism

Is either

- Toxicity to algae: $Log_{10}EC_{50}$
- Toxicity to crustaceans (Daphnia): Log10EC50
- Toxicity to fish: Log10LC50

F: > 2 (100 mg/L)

U: < - 2 (0.01 mg/L)

Manipulations:

- In the form of a decision tree

Variables considered, in order:

- Runoff and drift potential, Position, Field half-life, Aquatic toxicity
- Score from 0 to 1

3. Risk of groundwater contamination:

a Leaching potential

Based on Groundwater Ubiquity Score (GUS)

From Gustafson (1989, cited)

GUS based on: soil half-life and the pesticide mobility in the soil, as estimated from the octanol-water coefficient (K_{oc}): GUS = $\log_{10} (DT_{50}) * (4-\log_{10}(K_{oc}))$

F: < 1.8

U: > 2.8

b Position of application

F: 100% on crop

Membership in F and U: varies with % crop cover

U: 0% on crop

c Season (week) of application ("soil leaching risk")

F: week 22 to 34

U: week 1 to week 8 OR week 48 to 52

d Chronic toxicity to humans

Based on $\log_{10}$ (Acceptable daily intake)

F: > 0 (1 mg/kg per day

U: < -4 (0.0001 mg/kg per day)

Manipulations:

- In the form of a decision tree
- Variables considered, in order: GUS, Position, Leaching risk, Human toxicity
- Score from 0 to 1

4. Risk to air:

a Pesticide volatility

Based on Log_{10} of dimensionless Henry's Law Constant

F: < $\text{Log}_{10}\, 2.5 \times 10^{-6}$

U: > $\text{Log}_{10}\, 2.5 \times 10^{-4}$

b Position of application

F: 100% on crop

Membership in F and U: % applied in soil / 100

U: 0% on crop or soil

c Field half-life

(See above)

d Human toxicity

(See above)

Manipulations:

- Decision tree: Volatility, position, human toxicity
- Scores between 0 and 1

Other notes:

All modules can be considered individually or can be aggregated into an overall indicator estimating the potential environmental impact of a single pesticide application, again on a 0 to 1 scale.

New modules (e.g. risk to beneficial arthropods) can easily be added.

Roussel et al. (2000) have modified Ipest, applied to field crops : Ipest-B. The original model was modified to be applicable to hydrological conditions in Brittany (hence Ipest-B)

Full reference for this new model:

Roussel, O., Cavelier, A., van der Werf, H.M.G. (2000). Adaptation and use of a fuzzy expert system to assess the environmental effect of pesticides applied to field crops. Agriculture, Ecosystems and Environment 80:143-158.

CLM DUTCH ENVIRONMENTAL YARDSTICK FOR PESTICIDES

Netherlands
As reviewed by Levitan (1997) and CAPER

General notes:

- Serves as a tool for farmers but is also used to develop standards associated with green label incentives and has also been used to evaluate Dutch pesticide policy
- Environmental Impact Points (EIP) are presented to farmers in a workbook. Farmers chose the table appropriate to given site-specific conditions (e.g. crop, season, % organic matter…). From the tables, they select the score for the pesticides in use or those being compared
- Users chose which of three environmental indicators is more critical under their situation-specific conditions (groundwater, soil organisms, or aquatic organisms risk indicator)
- EIPs are calculated using this equation: EIP = (PEC / Maximum Predicted Concentration set by the Dutch government) * 100
- Points are initially calculated on the basis of an application of 1 kg a.i. per hectare. They are adjusted proportionally for other application rates by multiplying points by the recommended (or applied) application rate of active ingredient. To facilitate the use of the yardstick by farmers, the points for active ingredients are transformed into points for formulated products by multiplying the a.i. content of the product by the number of points for the active ingredient. When product contains more than one active, the points are summed.

Model structure:

- Is a decision tree
- EIPs stand alone and are not combined into an overall risk score

Elements of the model:

(i.e. requirements for PEC calculations)

1. Risk to groundwater

    Variables are:

- Pesticide soil degradation rate
- Mobility in soil (adsorption coefficient KOM)
- Dosage per ha
- Mobility in soil (soil organic matter content)
- Season of application – will impact degradation and mobility

2. Risk to water organisms

   (surface water)

   Variables are:

   - Acute toxicity (LC_{50}) of the most sensitive organism
   - Dosage per hectare
   - Method of application – will affect emission to surface water

   The application method is used to calculate an emission percentage:

   - 0% for granules and seed treatments;
   - 0.5% for sprays on rows;
   - 1% for full field spraying of arable crops;
   - 10% for full field spraying of fruits;
   - 100% for aerial spraying.

3. Risk to soil organisms

   Variables are:

   - Acute and chronic toxicity to soil organisms (LC_{50} and NOEC)
   - Soil degradation rate
   - Mobility in soil (K_{OM})
   - Dosage per ha
   - Mobility in soil (soil organic matter content)

   The PESTLA simulation model is used to calculate leaching potential and persistence under Dutch soil conditions

CONSUMER UNION'S INDICES OF TRENDS IN AGRICULTURAL PESTICIDE RISK
As reviewed by Levitan (1997)

General notes:

To assess weather regulatory policies have succeeded in reducing pesticide risk since the US Federal Insecticide, Fungicide and Rodenticide Act (FIFRA) was revised in 1972.

Model structure:

Algebraic (i.e. no categorical indices, not a composite index)

Elements of the model:

1. Relative acute toxicity to humans

   Beyond the scope of NAESI - although manipulations are interesting:

   Use-weighted acute toxicity average:

   $$\frac{\text{(pounds used * rodent LD}_{50}\text{) for every active ingredient in a given class of pesticide}}{\text{total pounds used (i.e. for all a.i. considered) for that class of pesticide}}$$

   Use-weighted toxicity also derived for the median and most toxic and least toxic decile of each class of pesticide. The most toxic decile for herbicides, for example, is the group of most toxic a.i. which account for 10% of the lbs applied. These results were used to calculate a toxicity differential between the most and the least toxic pesticide option. Divide the average LD50 of the least toxic decile by the average LD50 of the most toxic decile.

2. Relative chronic toxicity to humans

   Beyond the scope of NAESI

USDA ERS CHRONIC AND ACUTE RISK INDICATORS OF PESTICIDE USE

As reviewed in Levitan (1997)

General notes:

Based on “toxicity/persistence units” (TPU) which are calculated for each active ingredient. These are essentially toxicity weighted measures of pesticide use – instead of only using the amount used as a proxy for risk.

However, this model has only addressed risk to humans.

SYNOPS 2

Germany
Gutsche and D. Rossberg (1998)
Also reviewed by Levitan (1997) and CAPER

General notes:

- To monitor risk trends since the German Plant Protection Act was amended
- Although at a national level, it incorporated site-specific data – made assumptions about typical soil and water conditions in the nation, treating arable land as a single field site.
- Does not include a ground water since a pesticide which has problems concerning leaching to groundwater would not be registered in Germany. Compartments considered are soil, surface water, and air (optional).
- Spray only; not risks from seed treatments or non spray techniques

Model structure:

TER are calculated for different taxa. Results are not combined. These are presented in a 'risk graph', divided in 4 sectors (earthworm, daphnia, fish, and algae) and each is divided in acute and chronic risk. The index value determines the arc radius of the segment; the larger the sector, the larger the risk.

Elements of the model:

Steps are:

1. Define the pest management strategy

    Each strategy has a fixed number of a.i applications (may be repeated applications of a single a.i.) for which rate of application (label) and application time are related to the developmental stage of the crop (BBCH code). Seed treatments and other non-spray techniques are not considered.

    Variables are:

    - BBCH code to determine the Application rate
    - Application time (days)

2. Calculate the predicted environmental concentration (PEC) for each application of the strategy and for each compartment, in the short term (direct load) and long term:

    a Direct soil load = (application rate – drift) * (plant-soil surface distribution)

    - In mg/kg of soil

- Distribution between the plant surface and soil surface depends on the crop stage
- Drift = application rate (g a.i./ha)* spray drift value (%)
- Spray drift value is dependant on the crop and the distance between sprayer and the surface water body; from the Ganzelmeier table (Ganzelmeier 1997, cited). Default value for distance is 5m.

b Direct water load = drift * water index

- In mg/L water
- Water index (in %) is based on the proportion of field lengths that border a water body in a region – the proportion of fields neighbouring a water body can serve as a first estimate.

c The long term environmental concentration is a function of degradation rate and pesticide absorption to soil or sediment particles. It is calculated over a one year span (→ peeks on a 365 day graph). Data for organic content, depth to water table, soil movement, temperature, etc. are based on the aforementioned assumptions about typical conditions. Input variables such as DT_{50}, water photolysis, and hydrolysis are considered.

Concentrations over time are first calculated for single applications, and then they may be added.

Units for long-term PEC are mg*d/kg of soil and mg*d/L of water

3. Calculate the biological risk

Is calculated as an acute and chronic risk to earthworms, aquatic invertebrates, algae and fish, using previously established PECs for every a.i. application. Risk to earthworms draws on calculations for PEC in soil, while risk to daphnia, algae, and fish draws on PEC for water. Calculates the ratio between exposure and toxicity:

a Endpoints for acute risk are:

- LC_{50} earthworm, daphnia, algae, and fish

Manipulations:

- short-term PEC for appropriate compartment / LC_{50}

b Endpoints for chronic risk is the NOEC

Manipulations:

- long-term PEC for appropriate compartment * test duration / NOEC

There is a multiplication by the duration of NOEC experiment so that calculations lose their time dimension. Usually the time is standardized (14 d for earthworms, 4 d algae, and 21 days for daphnia and fish) but may also deviate from the standard.

Due to sparse data, bioconcentration factors were estimated using QSAR

- For fish: $\log BCF_{fish} = 0.99K_{ow} – 1.47 \log (4.79 \times 10^{-8} \times K_{ow} + 1)$
- From Nendza (1991, cited)
- For earthworms: $\log BCF_{ew} = 1.098 \times \log K_{ow} – 22917$
- From Pflugmacher (1992, cited)

SYNOPS also intends to calculate a food chain risk on the basis of the NOEL and bioconcentration factors for birds and mammals.

U. CALIFORNIA ENVIRONMENTAL HEALTH POLICY PROGRAM RANKING SYSTEM

As reviewed by Levitan (1997)

General notes:

- focus on the most hazardous pesticides; to influence pesticide risk reduction policy in California

Model structure:

- 12 indicators are considered independently and are given a rating. These are based on the statistical distribution of data for each variable. Pesticides in the top 10% (e.g. lowest 10% of LD50 value or highest incidence of observations in groundwater) receive a score of 4; score of 3 for the 75-90 percentile; score 2 for the 50-75 percentile; score of 1 for the 0-50 percentile. Greater discrimination is made at the high end of the scale because of greater interest in the most hazardous group of pesticides.
- The summary hazard index is an additive function

Elements of the model:

1. Human health

    4 variables are scored from 1 to 4

    These indices are beyond the scope of NAESI

2. Ecological health

    Each variables is scored from 1 to 4:

    a Avian LD_{50} (→ oral?)

    b Invertebrate LC_{50} (→ terrestrial or aquatic?)

    c Fish LC_{50}

    d Bioconcentration factor

3. Natural resources

    a Solubility in water

    Rating from 1-4

    b Ground water contamination: actual well detections in California

    Rating from 0-4

    c Soil adsorption

Rating from 1-4

d Field half-life

Rating from 1-4

Manipulations:

Weighting factors are assigned to the 3 broad categories but within those categories, all variables are weighted equally:

Total hazard index value =

0.7 (sum of human health ratings) + 0.2 (sum of environmental health ratings) + 0.1 (sum of natural resources ratings)

Data gaps filled by using median

Reference:

Pease, W.S., J. Liebman, D. Landy, and D. Albright (1996). Pesticide Use in California: strategies for reducing environmental health impacts. An environmental health policy program report, Center for Occupational and Environmental Health, School of Public Health, California Policy Seminar, University of California, Berkeley, California, USA, 116 pages

THE HASSE DIAGRAM (HD)

Denmark
Reviewed by CAPER

General Notes:

- - To rank pesticides, mainly in relation to their risk to groundwater.
- - Unit of analysis: active ingredient
- - No scores
- - Generates a network diagram where the ranking is not linear - the ranking is relative: pesticides can only be ranked in relation to each other while no information can be given on a single substance (as opposed to an absolute ranking gives a final number but where contradictions between variables are hidden).

Method:

1. Chose the variables (these are not fixed; the papers cited below use mostly physico-chemical variables such as half-life, water solubility, vapour pressure, and also usage variables such as dose and sprayed area)
2. Build a matrix with appropriate data
3. The Hasse Diagram is in fact a model, which generates a graphical result

Reference:

Halfon, E., S. Galassi, R. Brüggeman, and A. Provini (1996). Selection of Priority Properties to Assess Environmental Hazards of Pesticides. Chemosphere 33(8):1543-1562

Sørensen, P.B., B.B. Mogensen, S. Gyldenkærne, and A.G. Rasmussen (1998). Pesticide Leaching Assessment method for Ranking Both Single Substances and Scenarios for Multiple Substance Use. Chemosphere 36(10):2251-2276

ENVIRONMENTAL PERFORMANCE INDICATORS FOR PESTICIDES (P-EMA)

U.K.
Brown et al. (2003); Hart et al. (2003)
Also reviewed by CAPER

General notes:

- Part of a holistic computer-based system known as EMA (Environmental Management for Agriculture) to quantify 'best practice' at a farm level
- The model allows for adjustments according to site-specific conditions and agricultural practice

Model structure:

- A previous version of the system was based on warning phrases which appeared on pesticide labels.
- A recent version based on TERs and a HQ for honey bees, as proposed by the EU Uniform Principles
- Risk is scored and aggregated into final index

Elements of the model:

Exposure is addressed in Brown et al. (2003)

1. Acute risk to birds

   $TER_{avian\_acute} = LD_{50}$ / avian acute exposure

   Where avian acute exposure = (c * f)/w

   - $c_{treated\ seeds}$: nominal concentration (mg a.i. / kg)
   - $c_{pellets}$: nominal concentration (mg a.i. / kg)
   - $c_{spray\ and\ granules}$ = a * r (in mg/kg food) where
     (a) a: application rate (kg/ha)
     (b) r: residue factor ((mg/kg food)/kg/ha)
   - f_{spray}: daily food intake (wet weight in kg), (assumed 72% moisture in insects and 13% in seeds
   - $f_{granules}$: 69 * average granule weight, (assumed 69 particles are ingested and lost every day)

- w: body weight in kg

To reflect the variation in bird behaviour, two versions of the acute avian exposure can be calculated for a bird feeding entirely in the crop (centre-feeding bird) and for a bird feeding only partly in the crop (edge-feeding bird):

Avian acute exposure = $(c_{crop} * f_{crop} + c_{drift\ zone} * f_{drift\ zone})$ /w

- c_{crop}: residues on food from crop (mg a.i. / kg)
- c_{drift} zone: residues on food from drift zone (mg a.i. / kg)
- f_{crop}: amount of food taken from crop (kg)
- $f_{drift\ zone}$: amount of food taken from hedges and conservation headlands (kg)
- Suggested scenarios for indicator species are reported:
  (a) For sprays in arable crops and on orchard trees and fruit bushes: 11g blue tit consuming 11 small insects / day
  (b) For sprays on the ground under orchards and fruit bushes: 18g European robin consuming 17g small insects / day
  (c) For pellets and treated seeds: 22g tree sparrow consuming 6.3 pellets or treated seeds / day

See scoring below

Note that for birds, only the lowest score between acute and short-term is preserved

2. Short-term risk to birds

   $TER_{short\_term} = LC_{50} / c$

   This equation applies to center-feeding birds. For edge-feeding birds, the short-term TER is replaced by the weighted average concentration for food obtained in three areas: crop, hedge/headland, and natural habitat.

   Short-term TER not calculated for granules

   Weighting factors can be added:

   - Incorporation of treated seeds, granules or pellets into the soil leads to reduced exposure by a factor of 10.
   - If spoil removal is included, the exposure is reduced by a factor of 100.

   See scoring below

3. Acute risk to mammals

   TER = LD_{50} / mammal acute exposure

   Where mammal acute exposure = (c * f)/w

   - $c_{treated\ seeds}$: nominal concentration (mg a.i. / kg)
   - $c_{pellets}$: nominal concentration (mg a.i. / kg)

- $c_{spray\ and\ granules}$ = a * r (in mg/kg food) where
  (a) a: application rate (kg/ha)
  (b) r: residue factor ((mg/kg food)/kg/ha)
  (c) r is 112 for mammals eating short grass in arable crops
- f_{spray}: daily food intake (wet weight in kg), (assumed 77% moisture in short grass and 13% in seeds)
- $f_{granules}$: if a single granule contains a LD, then the p-EMA score is set to its maximum value of -100.
- w: body weight in kg

Suggested scenarios for indicator species are reported:

- For sprays in soft fruit: 25g field vole consuming 14g short grass / day
- For sprays in arable crop (centre-feeding mammal): 3.33kg hare consuming 803g short grass / day
- For sprays in arable crop (edge-feeding mammal): 25g field vole consuming 14g short grass / day
- For sprays in orchards: 1.5kg rabbit consuming 417g short grass / day
- For pellets and treated seeds: 18g wood mouse consuming 2.9g seeds or pellets / day

In orchards, when there is a strip of soil under each row of trees that is kept clear or with few weeds, then the exposure is reduces by a factor of 10. Weighting factors can also apply for soil incorporation and spoil removal, as the case for birds

See scoring below

4. Acute risk to earthworms

$TER_{earthworm\_acute} = LC_{50}$ / initial soil concentration

Where the initial soil concentration is obtained through modelling (see Brown et al. 2003)

See scoring below

5. Acute risk to bees

Hazard ratios (HQ) are calculated:

- For spray: HQ = application rate / LD_{50}
- For seed treatments, granules and pellets, exposure is considered negligible: HQ = 0
- For applications between October and February: HQ = 0
- For non-flowering crops (e.g. corn): HQ = 0

- When low exposure but flowering: HQ = proportion of application rate deposited as drift / LD_{50} (drift estimates are obtained through modelling – see Brown et al. 2003)
- When exposure is low in crop + margin: HQ = 50
- If HQ < 2500 but the a.i. has significant insect growth regulator activity, then HQ = 2500 (high hazard)

See scoring below

6. Risk to non-target arthropods

   Not based on a risk quotien approach; simply scored

   - Score 0: no insecticidal activity or solid formulations or seeds
   - Score -50: Selective insecticides or ICP insecticides
   - Score -90: Active against a broad spectrum of insects
   - Score -80: Other insecticides or other pesticides with insecticidal activity

7. Risk to aquatic organisms

   TER = toxicity/PEC

   Acute TER calculated for fish, Daphnia, algae and Lemna

   Chronic TER calculated for fish and Daphnia

   PEC based on Application rate

   - Soil type
   - Soil organic matter
   - Crop cover
   - Method of application

Scoring (except for non-target arthropods) is in two steps:

- Identify threshold values for each risk index. These will distinguish among three risk categories (average good practice, below average/review recommended and poor). These values are based on thresholds currently used in regulatory assessments (e.g. if the TER is below 10); table III in Brown et al.
- Draw a straight line through the two threshold values to define scores for the other values. These lines can give risk scores ranging from minus infinity to plus infinity, but limits were set at 0 and -100. These scores is an indication of the negative impact that may be caused by a given treatment. From 0 to -40 indicates 'a general good practice', from -40

to -70 indicates ‘below average/review recommended’, and -70 to -100 indicates a ‘poor’ agricultural practices.

ENVIRONMENTAL POTENTIAL RISK INDICATOR FOR PESTICIDES (EPRIP)

Italy
As reviewed by CAPER

General notes:

- Is used as a tool for farmers
- Includes site-specific data

Model structure:

- Based on a risk quotient appraoch
- Results are scored and aggregated (multiplied)

Elements of the model:

1. Risk to humans

   Beyond the scope of NAESI

2. Risk to earthworms

   PEC_{soil} / LC50 $_{earthworms}$ in mg/kg dry soil

   After a single application:

   PEC_{soil} = aprate * (1 - f_{int}) / (100 * mixing depth * soil bulk density)

   For many applications:

   $PEC_n = PEC_{soil} * (1 – exp^{-nki}) / (1- exp^{-ki})$

   - n is the number of applications
   - k is the dissipation rate = ln2/soil half-life
   - i is the number of days

   See scoring below

3. Risk to algae, crustaceans, and fish in surface water by drift

   PEC_{drift} / LC50 $_{water\_organisms}$

   Where PEC_{drift} = aprate *f_{drift} / volume of water in the ditch

   See scoring below

4. Risk to algae, crustaceans, and fish in surface water by run off

   PEC_{runoff} / LC50 $_{water\_organisms}$

The amount of pesticide translocated into surface waters depends on the slope, soil texture, intensity of the rain event, the distance between the treated area and the ditch, and on the elapsed time between pesticide application and onset of rainfall.

See scoring below

The highest score (drift vs. runoff) is used in the final index manipulations

5. Risk to human through volatilization

Beyond the scope of NAESI

Scoring criteria:

All risk values are normalized in a scale from 1 to 5

- Score 1: risk value < 0.01
- Score 2: risk value < 0.1
- Score 3: risk value < 1
- Score 4: risk value < 10
- Score 5: risk value > 10

Manipulations:

EPRIP =

$Score_{groundwater} * \max(Score_{surface\_water\_by\_drift}, Score_{surface\_water\_by\_run\_off}) * Score_{soil} * Score_{air}$

Final index between 1 and 625

SYSTEM FOR PREDICTING THE ENVIRONMENTAL IMPACT OF PESTICIDES (SYPEP)

Belgium
As reviewed by CAPER

General notes:

- To inform farmers, extension services and regulating authorities about the environmental impact of pesticides in a certain region.

Model structure:

- Calculation of Toxicity Exposure Ratio (TER) for groundwater and surface water; for groundwater the drinking water standard of 0.1 µg/L is used; for surface water short term and long term risks for water organisms are distinguished, based on the Maximum Permissible Concentration (MPC).

Elements of the model:

Prior to TER calculations:

- Estimation of the amount of pesticides reaching the soil in the area of concern;
- Prediction of the concentration in groundwater;
- Prediction of the concentration in surface water caused by spray drift, run off, drainage and pesticide handling;
- $TER_{groundwater} = 0.1 / PEC_{groundwater}$
- $TER_{surface\_water-short\_term} = MPC * 10 / PEC_{surface\_water-short\_term}$
- $TER_{surface\_water-long\_term} = MPC / PEC_{surface\_water-long\_term}$

TER values are transformed into score between 0 and 5 and combined to a total score between 0 and 15:

$SyPEP = Score_{groundwater} + Score_{surface\ water-short\ term} + Score_{surface\_water-long\_term}$

PESTICIDE ENVIRONMENTAL RISK INDICATOR (PERI)

Sweden
As reviewed by CAPER

General notes:

- Trends in indicator values over a number of years evaluates pesticide use by farmers as part of a certification process

Model structure:

This system comprises of 3 indicators:

- PERI-Handling: to assess the farmers' attitude toward pesticides (e.g. handling of pesticides, education, record keeping, etc.).
- PERI-Dose: to determine the dose used (i.e. actual dose relative to the recommended dose times the sprayed area in relation to field area in rotation).
- PERI-Environmental contamination and non-target organisms: shows ecological risk.

Elements of the model:

For ecological risk:

1. Risk of ground or surface water contamination

    Based on GUS

    - Score 1: < 0
    - Score 2: 0 - 1
    - Score 3: 1.0 - 1.8
    - Score 4: 1.8 - 2.8
    - Score 5: > 2.8

2. Risk of vaporization into the air

    Based on K_{aw} or Henry's Law Constant

    - Score 1: K_{aw} < 0.0001 or Henry's Law Constant < 1
    - Score 2: K_{aw} = 0.0001 – 0.0003 or Henry's Law Constant = 1 – 5
    - Score 3: K_{aw} = 0.0003 – 0.01 or Henry's Law Constant = 5 – 25
    - Score 4: K_{aw} = 0.01 – 1 or Henry's Law Constant = 25 – 100
    - Score 5: K_{aw} > 1 or Henry's Law Constant > 100

3. Risk of soil contamination

Based on acute toxicity to indicator organisms:

- An average of the toxicity scores for available species is calculated

Scoring criteria for Daphnia, Scenedesmus, or Chorella. Based on LC_{50} or EC_{50} in mg/L:

- Score 1: > 100
- Score 2: 10 – 100
- Score 3: 1 – 10
- Score 4: 0.1 – 1
- Score 5: > 0.1

Scoring criteria for earthworms. Based on LC50 in mg/kg dry soil:

- Score 1: > 1000
- Score 2: 1000 – 100
- Score 3: 10 – 100
- Score 4: 1 – 10
- Score 5: < 1

Toxicity scores for bees, based on bee acute oral toxicity in mg/bee

Scores are the same as for Daphnia etc.

4. Risk of bioaccumulation

Based on K_{ow} or BCF

- Score 1: Kow < 3.0 or BCF < 100
- Score 3: BCF = 100 - 1000
- Score 5: Kow >= 3 or BCF > 1000

Manipulations:

(GUS score * Kaw score + (Mean toxicity score * Kow score)/10)

- The outcome is an index between 2.2 and 7.5
- The environmental index can be multiplied with the dose indicator (although not discussed here)

BEES RISK INDICATOR (BRI)

Italy
Villa et al. 2000

General notes:

- Risk for honey bees from exposure to pollen
- Authors underline the ecological importance of this model – as pollen has the possibility of exposing the whole colony

Model structure:

- Based on hazard quotients
- Values are scored

Elements of the model:

For short-term risk:

TER ingestion: $$\frac{LD_{50\_oral}}{\text{Pollen ingestion PEC}}$$

TER contact: $$\frac{LD_{50\_contact}}{\text{Pollen contact PEC}}$$

1. PECs are set according to the probability of pollen contamination. An exposure index was developed, which is based on:
    - whether the a.i. is capable of incorporating into pollen, which is estimated with the log K_{oa} (K_{oa} being the octanol-air partition coefficient – ratio between K_{ow} and K_{aw}),
    - the amount applied and
    - the persistence of the chemical.
2. Equations:
    - Exposure index = Score(TWA) * Score(log K_{oa})
    - Time weighted average (TWA) = application rate * $(1 – e^{-kt})/kt$
      (a) Where t is time
      (b) And $k = \ln 2 / DT_{50}$
3. For multiple applications:

TWA = [application rate for first application * $(1 - e^{-kt})$ + application rate for subsequent application * $(1 - e^{-k(t-t1)})$] / kt

Scoring:

- Score 0.1: TWA < 10 g/ha
- Score 0.2: TWA = 10 – 20 g/ha
- Score 0.5: TWA = 20 – 50 g/ha
- Score 1: TWA = 50 – 100 g/ha
- Score 2: TWA = 100 – 200 g/ha
- Score 4: TWA = 200 – 400 g/ha
- Score 8: TWA = 400 – 800 g/ha
- Score 16: TWA = 800 – 1500 g/ha
- Score 32: TWA > 1500 g/ha

$K_{oa} = K_{ow}/K_{aw} = C_o/C_w * C_w/C_a = C_o/C_a$

Where

- C_o: saturation concentration (v/v) in octanol
- C_w: saturation concentration (v/v) in water
- C_a: saturation concentration (v/v) in air
- K_{ow}: concentration octanol / water
- $K_{aw} = H/R*T$

  (a) H = Henry's law contant (Pa m^3/moles)

  (b) R = 8.314 Pa m^3/moles

  (c) T = absolute temperature (25°C)

Scoring:

- Score 0.1: log $K_{oa} < 5$
- Score 0.2: log $K_{oa} = 5 – 6$
- Score 0.5: log $K_{oa} = 6 – 7$
- Score 1: log $K_{oa} = 7 – 8$
- Score 2: log $K_{oa} = 8 – 9$
- Score 4: log $K_{oa} = 9 – 10$

- Score 8: log K_{oa} = 10 – 11
- Score 16: log K_{oa} = 11 – 12
- Score 32: log K_{oa} > 12

Pollen PEC (μg/g)

- = 100 when exposure index > 256
- = 10 when exposure index > 64
- = 1 when exposure index ≥ 8
- = 0.1 when exposure index ≥ 4
- = 0.01 when exposure index < 0.4

This first PEC classification is then refined into pollen PEC for:

- ingestion i.e. amount of pesticide ingested with food in a day and for
- contact i.e. amount of pesticide picked up with pollen in a day.

The amount of xenobiotic ingested in a day is estimated at 70 mg * PEC / 100 mg body weight for larvae and 5 mg * PEC / bee for adults. The amount of pollen collected in a day by a worker bee is assumed to be 300 mg * PEC.

Only toxicity data for adults are available; still used with larvae pollen ingestion PEC as an approximation for risk to larvae

RIVM INDICATOR

Netherlands
As reviewed in OECD Survey of National Risk Indicators, 1999-2000

General notes:

- To measure environmental risk over time in the Netherlands
- Was initially for terrestrial risk, but the equivalent has been developed for the aquatic ecosystems also

Model structure:

- Sum for all active ingredients of (PEC/TOX) * area weighted average

Elements of the model:

1. AARI: Acute Aquatic Risk Indicator for Pesticides

   PEC = PECditch = mean dosage (kg/ha) * 0.4 * mean fraction drift

   - Assume ditch with depth of 0.25m
   - Based on a model calculation which uses application rate and losses due to drift

   TOX is acute toxicity to aquatic organisms

   Variables are LC_{50} or EC_{50}

   Indices are calculated separately for Daphnia, fish and algae.

   Geometric mean of available data

   More recently, an indicator for groundwater was introduced. For the acute indicator for the groundwater ecosystem toxicity values for Daphnia are used as a surrogate.

   Area-weighted average = kg a.i. sold / mean dosage

   This gives the number of hectares treated with a.i. and is expressed as a fraction of the total number of hectares treated by all compounds.

2. ATRI: Acute Terrestrial Risk Indicator for Pesticides

   Also recently, two indicators introduced for the terrestrial compartment (ATRI): one based on earthworms, one on birds (none for mammals, only the partridge is used – because birds are generally more sensitive than mammals and because the partridge is known to forage in arable areas and pesticides are linked to its decline).

   The predicted environmental concentration is basically calculated from sale and area treated, but for earthworms, is modified according to soil properties. The calculations of risks to earthworms rely on field dose rates and toxicity values corrected for soil concentration of organic carbon. The daily intake of pesticides in partridges is calculated from the energy expenditure of a ‘standard-sized’ bird.

DANISH LOAD INDEX (DLI)

Denmark
As reviewed in HAIR (unpublished); also discussed in OECD (2004)

General notes:

- Its purpose is to clarify whether tracked changes were due to changes in sales data or to changes in the toxicity of a pesticide following re-evaluation.
- Calculated for each year – although sliding means for 3 years have been used for sales data and area to reduce the effect of extreme years.

Elements of the model:

DLI = sum for all active ingredients: sales in kg / (tox value * area of arable land in ha)

Where tox value = $\frac{\text{standard dose applied to crop or crop type}}{\text{toxicity endpoint (e.g. } LC_{50}, LD_{50}, EC_{50}\text{, NOEL, NOEC…)}}$

Can be calculated for many organisms (e.g. fish, algae, mammals, etc.) but only the lowest tox value (i.e. the most sensitive organism) is incorporated in the final indicator formula.

ESCORT_2

(European Standard Characteristic Of non-target arthropod Regulatory Testing)
EU
Candolfi et al. (2000); also in HAIR (unpublished)

General notes:

- Guidance for terrestrial non-target arthropod testing and risk assessment
- Was developed to address limitations of previous guidelines (e.g. use of limit testing and an arbitrary 30% threshold value)

Model structure:

- Based on a risk quotient approach; for in-field and off-field scenarios.

Elements of the model:

For Tier I risk assessment:

1. $HQ_{in\_field} = \frac{\text{application rate * MAF}}{LR_{50}}$

   Where the application rate is in g or mL / ha

   - LR_{50} is the application rate causing 50% mortality
   - MAF (multiple application factor): is derived from the half-life of the product, the spray interval and the number of applications

2. $HQ_{off\_field} = \frac{\text{app. rate * MAF * (drift factor / vegetation distribution factor) * correction factor}}{LR_{50}}$

   Drift factor: % drift based on the 90th percentile drift data / 100

   Vegetation distribution factor: usually a factor of 10 is assumed appropriate to correct the overestimated exposure given by the 90th percentile drift values but still estimate the worst-case deposit.

   Correction factor: a safety factor of 10 to account for the extrapolation from the indicator species used in first tier (Aphidus rhopalosiphi and Typhlodromus pyri) to all other non-target arthropods.

FREQUENCY OF APPLICATION (FA)

Denmark
As reviewed in HAIR (unpublished); also discussed in OECD (2004)

General notes:

- Used as an indicator of the general environmental impact due to pesticide use
- Sum per year

Elements of the model:

1. FA = sum for all active ingredients [(SA/SD)/AGRA]

   Where SA = quantity of each a.i. (kg) sold per year

   - SD = standard dose of each a.i. in each crop type (kg / ha)
   - AGRA = area of arable land for that year (ha)

   The final FA indicator may be multiplied by 1000 for convenience.

MULTI ATTRIBUTE TOXICITY FACTOR (MATF)

USA
Benbrook et al. (2002)

General notes:

- For eco-labelling; applies only to potatoes in Wisconsin

Model structure:

- Based on the principle of risk = hazard * exposure potential
- Toxicity is a score which is obtained by adding scores of different

Elements of the model:

1. Acute mammal toxicity

   Refers to human impacts

   Beyond the scope of NAESI
2. Chronic mammal toxicity

   Refers to human impacts

   Beyond the scope of NAESI
3. Leaching Index

   Drinking water exposure as a potential source of human health risk

   Beyond the scope of NAESI
4. Ecological toxicity (ECO)

   a Avian index: see the model developed by Mineau (below: Ref ID 25)

   b Fish index

   Based on LC_{50} for the rainbow trout and bluegill

   Manipulations:

   - Average of [(1/rainbow trout LC50) * scaling factor] and [(1/bluegill LC50) * scaling factor]

   c Invertebrate index

   - Based on Daphnia LC_{50}
   - 1/Daphnia LC50 * scaling factor

   ECO = scaled $1/LC_{50\ \text{daphnia}}$ + scaled $1/LC_{50\ \text{fish}}$ + scaled $1/LD_{50\ \text{bird}}$

Different scaling factors are used, depending on the pesticide, to narrow the wide range of index values: Higher values are scaled down to be within 2 times the standard deviation above the mean value for all pesticides.

5. Biointensive IPM (BioIPM)

   a Resistance

   Resistance score reflects the pesticide's likelihood of causing resistance in target pests in Wisconsin.

   - For insecticides: Score from 1 (less likely) to 3 (prone to develop resistance)
   - For fungicides: Score from 1 (remote chance) to 5 (likely to lead resistant phenotypes)

   b Impact on benificials

   Reflects the potential impact on non-target organisms in Wisconsin. Is based on the 'Toxic Effect' index:

   - Beneficial Impact Score = 100 / (5 – toxic effect)
   - Toxic effect values come either from EIQ (Kovach et al. 1992) or expert opinion.

   c Bee toxicity

   Bee toxicity score in a database from Pieter Oomen (→ to find!) which includes data on both contact and oral routes of exposure

   Bee toxicity = 10/bee toxicity

   Manipulations:

   - BioIPM = (Resistance score+ Beneficial impact score+ Bee toxicity score) * 0.05

6. Final manipulations:

   - Additional weight is added, depending on focus:

   a Wisconsin potatoes:

   MATF = [(0.5 * AM) + CM + ECO + (1.5 * BioIPM)] * application rate

   b Focus on human health:

   MATF = [(1.5 * AM) + (0.5 * CM) + (0.5 * ECO) + BioIPM] * application rate

   c Focus on environment:

   MATF = [(0.3 * AM) + (0.5 * CM) + (2.0 * ECO) + (2.0 * BioIPM)] * application rate

Reference:

Benbrook, C., D. Sexton, J. Wyman, W. Stevenson, S. Lynch, J. Wallendal, S. Diercks, R. Van Haren, and C. Granadino (2002). Developing a pesticide risk assessment tool to monitor progress in reducing reliance on high-risk pesticides. American Journal of Potato Research 79: 183-199

NORWEGIAN INDICATOR

Norway
Source: Pesticide risk indicators for health and environment – Norway. Norwegian Agricultural Inspection Service, 2000

General notes:

- As a policy/regulation tool – is used in Canada for product evaluation
- Can be used to monitor change over time.

Model structure:

- Based on TER and HQ calculations
- All indices are scored
- Final index obtained by adding all index scores

Elements of the model:

1. Health risk
    - Beyond the scope of NAESI
2. Risk to terrestrial organisms (T)
    a Risk to earthworms
    - Variables are:
        - Earthworm 14 day LC50
        - Application Rate (g/ha)
        - Fraction of pesticide intercepted by crop
        - Mixing depth (cm)
        - Dry soil bulk density (g/cm^3)
    -
    - Manipulations:
        - $TER_{earthworm} = LC_{50} / PIEC_{soil}$
        - $PIEC_{soil}$ = appl. rate * (1 - interception) / (100 * mixing depth * dry soil bulk density)
            - PIEC: immediately after spray
            - Interception is 0 when bare soil, or up to 0.5 when a crop is present (Fraction reaching soil = 1 – interception)

- Mixing depth is assumed to be 5 cm for application to the soil surface and 20 cm when incorporation
- 1.5 g/cm^3 is assumed for bulk density
- Using these assumptions, the concentration in soil immediately after a single spray application becomes:
- $PIEC_{soil}$ = appl. rate / 750 when no incorporation or interception
- = appl. rate / 1500 when no incorporation but 50% interception

= appl. rate / 3000 when incorporation but no interception

-
- Scoring:
  - Score 0: TER > 100
  - score 2: TER = 10-100
  - Score 3: TER < 10
- The European and Mediterranean Plant Protection Organization has defined a threshold value of TER for earthworms of 100, while the EU's Uniform Principles use a threshold of 10.

o Risk to bees

- Variables are:
  - Contact LD_{50} or Oral LD_{50} (µg/bee)
  - Application rate (g/ha)
-
- Manipulations:
  - HQ_{bees} = application rate / LD_{50}
- The highest HQ (i.e. chose between the HQ calculated with contact toxicity or that obtained with oral toxicity) is scored.
- Scoring:
  - Score 0: HQ <50
  - Score 0.5: HQ = 50 – 100
  - Score 1: HQ = 100 – 1000
  - Score 1.5: HQ = 1000 – 10000
  - Score 2: HQ > 10000
- EU's Uniform Principles use a threshold of 50.

    - Scores for bees can only go up to 2, whereas scores for some other terrestrial organisms can go up to 4. This is because products toxic to bees are labelled accordingly and exposure to bees will be reduced.
- Risk to birds
    - Variables are:
        - LC_{50}
        - Application rate (kg/ha)
        - RUD according to Hoerger and Kenaga (1972)
    -
    - Manipulations:
        - $TER_{bird} = LC_{50} / PIEC_{food}$
        - $PIEC_{food}$ = application rate * 30
            - RUD for leaves and small insects (RUD = 30) was chosen as a worst case scenario
    - If there is no diet data or if the diet data is unreliable, one can use the acute oral LD_{50} value in the TER calculation, taking into account the quantity of contaminated food the bird ingests. Assume that a small bird (10g) have a daily food intake of approximately 30% of their body weight, whereas a large bird (100g) will have a daily food intake of approximately 10% of their body weight. Equations become:
        - For small birds: $TER_{bird} = LD_{50}$ / application rate * 9
        - For large birds: $TER_{bird} = LD_{50}$ / application rate * 3
    - The constants 9 and 3 arise because the RUD of 30 is adjusted for the food intake per kg of body weight. This method assumes that birds ingest only contaminated food.
    - For treated seeds and granules, consider the concentration of active ingredient per granule (divide the toxicity found in the diet by the concentration of a.i.). If there is no diet data, calculate the consumption as above based on the $LD_{50.}$ Again, it is assumed that only contaminated seeds are ingested.
    -
    - Scoring:
        - Score 0: TER >10
        - Score 2: TER = 1-10
        - Score 4: TER < 1
    -

- The EU's Uniform Principles have defined a threshold value for TER of 10 in acute and sub-acute studies.
- If the product used has documented repellent effect, the TER is assumed to be zero.
-
- The score for overall risk of undesirable effects on terrestrial organisms (T) :
    - T = the highest of scores for earthworms, bees, or birds.

3. Risk to aquatic organisms (A)
    - Variables are:
        - LC_{50} or EC_{50} (for algae/water plants, Daphnia, and fish)
        - Spray drift
        - Surface runoff
    - $TER_{aquatic\ organism}$ = toxicity for aquatic organism / PEC
    -
    - PEC is based on spray drift and surface runoff (→ I'm not sure if different TER are calculated for different exposures or if both drift and runoff are included in the TER calculations)
    -
    - For spray drift, Ganzelmeier et al. (1995) is cited. Surface runoff is based on DT_{50}, K_{oc}, and solubility. Goss and Wauchope (1990) is cited.
    -
    - Scoring:
        - For acute studies with invertebrates or fish:
            - Score 0: TER > 100
            - Score 1: TER = 10 – 100
            - Score 2: TER = 1 – 10
            - Score 3: TER = 0.1 – 1
            - Score 4: TER < 0.1
    - The EU's Uniform Principles have defined a threshold value for TER of 100 for acute studies on Daphnia and fish.
        - For studies with algae or water plants:
            - Score 0: TER > 10
            - Score 1: TER = 10– 10

- Score 2: TER = 0.1 – 1
- Score 3: TER = 0.01 – 0.1
- Score 4: TER < 0.01

- The EU's Uniform Principles have defined a threshold value for TER of 10 for chronic trials and for trials on algae and water plants. However, to keep the system simple, chronic toxicity is not accounted for at this point.

4. Leaching potential (L)
   - Variables are:
     - GUS
     - Application rate
   - GUS is based on the chemical's adsorption (K_{oc}) and persistence in soil (DT_{50}):
     - $GUS = \log (DT_{50}) * (4 – \log(K_{oc}))$
   -
   - Scoring:
     - Score 0: GUS < 1.8
     - Score between 1.25 and 2: GUS = 1.8 – 2.8 (score varies according to application rate)
     - Score between 2.5 and 4: GUS > 2.8 (score varies according to application rate)
5. Persistence in soil (P)
   - Variables are:
     - DT_{50}
     - Application rate
   -
   - Scoring:
     - Score 0: DT_{50} < 10
     - Score between 0 and 1: DT_{50} = 10 – 30 (score varies according to application rate)
     - Score between 0.5 and 2: DT_{50} = 30 – 60 (score varies according to application rate)
6. Bioaccumulation (B)
   - Variables are:
     - BCF

- Pesticide persistance (DT_{50})
- Purification half-life

▪ DT_{50} is used for persistence because many pesticides lack degradation studies. BCF for a whole fish is used as a standard.

▪ Manipulations:
  - B = Score (BCF or $logP_{ow}$) * Score (DT_{50}) * Score (purification DT_{50})

▪

▪ Scoring:
  - Score 0: BCF = 0 or $logP_{ow} < 3$
  - Score 1: BCF = 100 – 1000 or $logP_{ow}$ = 3-4
  - Score 2: BCF > 1000 or $logP_{ow} > 4$

  - Score 0: DT_{50} < 1 day
  - Score 0.5: DT_{50} = 1 – 10 days
  - Score 1: DT_{50} = 10 – 60 days
  - Score 1.5: DT_{50} = 60 – 200 days
  - Score 2: DT_{50} > 200 days

  - Score 0: purification DT_{50} < 1
  - Score 0.5: purification DT_{50} = 1 – 10
  - Score 1: purification DT_{50} > 10

▪

Final environmental index:

$E = (T + A + L + 2P + B + 1)^2$

The number 1 is added so that pesticides with a score of zero for all indices are still taken onto account for total environmental risk. For treated seeds, the equation is modified:

$E = (2T_{bird} + A + L/2 + 2P + B + 1)$

It is assumed that for seed treatments, risk to bees and earthworms is null, but the risk to birds is increased so added weight is given to T_{bird}.

To monitor change over time:

- The environmental risk index (E) for each a.i. in each product is multiplied by the area (in decares : 0.1 ha) on which the product is used that year. These indices are added up:

Collective environmental risk index = E1area1 + E2area2 + … + Exareax

PESTICIDE OCCUPATIONAL AND ENVIRONMENTAL RISK (POCER)

Belgium
Vercruysse and Steurbaut (2002)

General notes:

- To evaluate the impact and risk of pesticides to both farm workers and the environment
- Based on the information required by the Uniform Principles

Model structure:

Final indicator is a score based on a method developed by Beinat and Van den Berg (1996 – a Netherlands Report, cited); describes the extent to which a chosen trigger is exceeded as a numerical dimensionless value:

- a- The lower and upper limits for each index are calculated. Since pesticides with a risk index lower than 1 meet the criteria set by the Uniform Principles, the lower limit is set to 1 (except for arthropods where it is 0). The upper limit is usually 100 (except for arthropods where it is 1 and for groundwater where it is 10000).
- b- The risk index, lower limit (except for arthropods), and upper limit are divided by the upper limit, and then transformed: log (1 + 1/result of division)
- c- The risk of a pesticide to the different components is related to the extent to which the lower limit is exceeded – formula for exceedence factor (EF) in ref. EF values lower or equal to 0 are set to 0 and indicate a low risk, EF values higher or equal to 1 are set to 1 and indicate a high risk. An intermediate risk is found for values between 0 and 1.
- d- For risk to environment, sum the values for the environmental components (i.e. assume that every component has equal importance)

Elements of the model:

For environmental risk:

7. Persistence in the soil
   - Based on the DT_{50}
   - In Uniform Principles: no authorization of a plant protection product is granted if the DT_{50} of the pesticide in soil is more than 90 days
   - In the Netherlands: no authorization is granted if the DT_{50} > 180 days

- Risk is set as a power equation in order to become a lower limit of 1 (for a DT_{50} of 90 days) and an upper limit of 100 (for a DT_{50} of 180 days):
- $RI_{persistence} = 10^{(DT50/90-1)\ x2}$

8. Risk to groundwater contamination
   - $RI_{groundwater}$ = PEC / 0.1
     - PEC calculated with the PETLA model
     - According to Uniform Principles the concentration of a.i. in water < 0.1µg/L
9. Acute risk to aquatic organisms
   - Variables are:
     - Fish LC_{50} or Daphnia EC_{50} or algae NOEC (see below)
     - Application rate (kg a.i./ha)
     - % drift deposition
     - Depth and width of ditch
   - For agricultural conditions in The Netherlands and Belgium, the exposure of aquatic organisms is mainly caused by pesticide spray drift; other exposure routes such as surface run-off and leaching are considered negligible.
   - $RI_{aquatic}$ = PEC / toxicity
   - PEC = (application rate * drift * width of ditch) / (width ditch * depth ditch * 1000)
     - Assume depth of 0.3m and width of 1m
     - Drift values are derived from Ganzelmeier et al. (1995).
     - The factor 1000 is a conversion factor for the units.
   - For toxicity, the lowest of these is used:
     - LC50 for fish/100
     - EC50 for Daphnia/100
     - NOEC for algae/10
   - Safety factors are those defined in Uniform Principles
   - Risk is considered negligible when application with treated seed, granules, dipping a plant in a pesticide solution or pouring a pesticide solution to a plant
10. Acute risk to birds
    - Variables are:
      - LD_{50}
      - Body weight

- Application Rate
- Granule weight

▪ Risk Index $_{birds}$ = (PEC_{bird} * 10) / (LD_{50} * body weight)

- 10 is the criteria set by Uniform Principles

▪ PEC calculations:

- For sprayed crop:
  - PEC_{bird} = 31 * application rate * body weight * 0.3
    31 from Hoerger and Kenaga (1972)
    0.3 because small birds eat 30% of their weight a day
    Default value for body weight is 0.01kg
- For seed treatment:
  - PEC_{bird} = application rate * body weight * 0.3
    Application rate in mg a.i./kg treated seed
    Default value for body weight is 0.01kg
- For granules:
  - PEC_{bird} = 20 * granule weight * fraction of a.i. in granule
    Consumption of 20 grit particles a day
    Default value for granule weight is 2mg

11. Acute risk to bees

▪ Variables are:

- LD_{50} (µg/bee)
- Application rate

▪ Manipulations:

- Risk Index $_{bees}$ = application rate / (LD_{50} * 50)
- Use the minimum between the oral and contact LD_{50} value
- 50 is the criteria set by Uniform Principles

12. Acute risk to earthworms

▪ Variables are:

- LC_{50} (mg/kg soil)
- Application rate (kg/ha)
- Depth of soil layer (m)
- Density of soil (kg/m^3)

- Fraction of spray reaching the soil (1 – interception factor)

▪ Risk Index $_{\text{earthworms}}$ = (PIEC * 10) / LC_{50}

- 10 is criteria set by Uniform Principles
- PIEC = (100*application rate*fraction reaching soil) / (depth * density)
  - Fraction reaching soil only applies to spray. Otherwise, it is assumed that 100% pesticide reaches the soil. Interception factors in cited reference.
  - Default value for depth of soil layer is 0.05m
  - Default value for density of soil is 1400 kg/m^3

13. Risk to beneficial arthropods

▪ If pesticide not sprayed, then Risk Index $_{\text{beneficial arthropods}}$ = 0

▪ If pesticide sprayed:

- Risk Index $_{\text{beneficial arthropods}}$ = (% reduction of control capacity – 25) / (100 – 25)

% reduction in control capacity (i.e. reduction in natural enemy potential or effectiveness, which if reduced could lead to higher pest numbers) refers to effects such as mortality and non-hatching of eggs and pupae and to sublethal effect such as reduced fertility or problems with regard to moulting, repellency, etc.

Based on online database of Koppert (→ find?) and the database of Biobest Biologcal Systems, Belgium. Effects are classified into 4 categories. Class 1 refers to a mortality less than 25%, class 2 refers to a mortality between 25 and 50%, class 3 to a mortality between 50 and 75% and class 4 to a mortality between 75 and 100%. For the POCER indicator, the arithmetic mean of the class of the beneficial arthropod with the highest mortality is used.

PESTICIDE RISK INDEX FOR HYPOGEAN OR EPIGEAN SOIL SYSTEMS AND FOR SURFACE WATER SYSTEMS

Italy
Finizio et al. (2001)

General notes:

- Based on the information required by the Uniform Principles
- Developed models for three different environments, at two different time scales

Model structure:

- Results form each index are scored according to negligible, low, medium, high, and very high risk – thresholds are cited in reference
- Scores are added for final manipulations – weights change depending on the model

Elements of the model:

A. FOR HYPOGEAN SOIL SYSTEMS (I.E. BELOW GROUND)

Short-term scale (PRIHS-1)

This index calculates the risk for non-target hypogean organisms immediately after a pesticide application. PEC is calculated assuming that the product spreads uniformly on a surface of 1 ha and on a layer of 5 cm. Assuming the density of soil to be equal to 1.5 g/cm3, the PEC can be calculated as:

- $PEC_{short\text{-}term}$ = MRA / 750
- Where MRA is maximum rate of application (g/ha)
- 750 = 10,000 m2 * 5 cm * 1.5 g/cm3 = 750,000 kg. As the PEC is expressed as milligrams per kilogram of soil, this value is corrected by a factor of 1000.

1. Short-term risk to earthworms
    - Variables are:
        - EC_{50}
        - Maximum application rate (g/ha)
        - Depth (cm)
        - Soil density (g/cm^3)

- Manipulations:
  - $TER_{ew\_short\text{-}term} = EC_{50} / PEC_{short\text{-}term}$
- Scoring:
  - Score 0: $TER_{ew\_short\text{-}term} > 1000$
  - Score 1: $TER_{ew\_short\text{-}term} = 100 – 1000$
  - Score 2: $TER_{ew\_short\text{-}term} = 10 – 100$
  - Score 4: $TER_{ew\_short\text{-}term} = 1 – 10$
  - Score 8: $TER_{ew\_short\text{-}term} < 1$

2. Short-term risk to beneficial arthropods
   - Variables are:
     - Maximum application rate (g/ha)
     - Inhibition of activity (%)
   - Scoring:
     - Score 0: level of inhibition is null at (2 * max application rate)
     - Score 2: level of inhibition is between 0% and 30% at the max application rate
     - Score 4: level of inhibition is > 30% at the max application rate
     - Score 8: level of inhibition is > 30% at (0.5 * max application rate)
3. Short-term risk to mammals
   - (→ Burrowing mammals?)
   - Variables are:
     - Dermal LD_{50}
     - Maximum application rate (g/ha)
     - Depth (cm)
     - Soil density (g/cm^3)
   - $TER_{mammal\_short\text{-}term} = LD_{50} / PEC_{short\text{-}term}$
     - Where $PEC_{short\text{-}term}$ is calculated as above
   - Scoring:
     - Score 0: $TER_{mammal\_short\text{-}term} > 1000$
     - Score 1: $TER_{mammal\_short\text{-}term} = 100 – 1000$
     - Score 2: $TER_{mammal\_short\text{-}term} = 10 – 100$
     - Score 4: $TER_{mammal\_short\text{-}term} = 1 – 10$

- Score 8: $TER_{mammal\_short\text{-}term} < 1$

▪ Final calculations for short-term risk (PRIHS-1 Index):

- PRIHS-1 = [5.5 * Score(Earthworm)] + [5 * Score(Beneficial)] + [2 * Score(Mammal)]
- Final score ranges from 0 to 100

▪ A low weight was assigned to mammals (for cutaneous exposure), assuming that their ecological role is relatively low in the hypogean system.

Long-term scale: PRIHS-2

1. Long-term risk to earthworms

▪ Variables are:

- 14-day NOEC
- Maximum application rate (g/ha)
- Depth (cm)
- Soil density (g/cm^3)
- Time of the toxicity test
- Soil half-life (DT_{50})

▪ Manipulations

- TERearthworm_long term = NOEC / PEClong term
  - Where $PEC_{long\ term} = PEC_{short\text{-}term} * ((1 - e^{-kt}) / kt)$
  - $PEC_{short\text{-}term}$ calculated as above
  - t: time of the toxicity test
  - k: $\ln 2/DT_{50}$

▪ Scoring:

- Score 0: $TER_{ew\_\ long\ term} > 1000$
- Score 1: $TER_{ew\_\ long\ term} = 100 - 1000$
- Score 2: $TER_{ew\_\ long\ term} = 10 - 100$
- Score 4: $TER_{ew\_\ long\ term} = 1 - 10$
- Score 8: $TER_{ew\_\ long\ term} < 1$

2. Long-term risk to micro-organisms

▪ Variables are:

- Maximum application rate (g/ha)
- Inhibition of activity (%)

- Microorganisms, not considered in the short-term index, have been included, assuming that their role is higher in the long run.
- Scoring:
  - Score 0: level of inhibition is null at (2 * max application rate)
  - Score 2: level of inhibition is between 0% and 25% at the max application rate
  - Score 4: level of inhibition is > 25% at the max application rate
  - Score 8: level of inhibition is > 25% at (0.5 * max application rate)

3. Long-term risk to arthropods
   - Variables are:
     - Maximum application rate (g/ha)
     - Inhibition of activity (%)
   - Scoring:
     - Score 0: level of inhibition is null at (2 * max application rate)
     - Score 2: level of inhibition is between 0% and 30% at the max application rate
     - Score 4: level of inhibition is > 30% at the max application rate
     - Score 8: level of inhibition is > 30% at (0.5 * max application rate)
4. Long-term to mammals
   - Variables are:
     - 2-yr NOEL
     - Maximum application rate (g/ha)
     - Depth (cm)
     - Soil density (g/cm^3)
     - Time of the toxicity test
     - Soil half-life (DT_{50})
     - Bioconcentration factor
   - Exposure via contaminated food; in this case a diet concentration (DC: mg/kg), expressed as the product of the bioconcentration factor (BCF) and the $PEC_{long\ term}$, has been calculated:
   - Manipulations:
     - $TER_{mammal\_long\ term}$ NOEL / (BCF * $PEC_{short\ term}$)
   - Scoring:
     - Score 0: $TER_{mammal\_\ long\ term} > 1000$

- Score 1: $TER_{mammal\_ long\ term}$ = 100 – 1000
- Score 2: $TER_{mammal\_ long\ term}$ = 10 – 100
- Score 4: $TER_{mammal\_ long\ term}$ = 1 – 10
- Score 8: $TER_{mammal\_ long\ term} < 1$

▪ Final calculations for long term risk (PRIHS-2 Index):

- PRIHS-2 = [4 * Score(Earthworms)] + [4 * Score(Micro-organisms)] + [3 * Score(Arthropods)] + [1.5 * Score(Mammal)]

B. FOR EPYGEAN SOIL SYSTEM

Short-term scale: PRIES-1

1. Short-term risk to bees

▪ Variables are:

- LD_{50} (µg/bee)
- Maximum rate of application (g/ha)

▪ Manipulations:

- HQ = maximum rate of application / LD_{50}

▪ Scoring:

- Score 0: HQ < 1
- Score 1: HQ = 1 – 10
- Score 2: HQ = 10 – 100
- Score 4: HQ = 100 – 1000
- Score 8: HQ > 1000

2. Short-term risk to birds

▪ Variables are:

- LD_{50}
- Total Daily Intake (mg/kg)

▪ $TER_{bird\_short\ term} = LD_{50}$ / TDI

- TDI is calculated on the basis of the concentrations typically reached on crops after a pesticide treatment, evaluated according to Hoerger and Kenaga (1972).

▪ Scoring:

- Score 0: $TER_{bird\_short\ term} > 1000$
- Score 1: $TER_{bird\_short\ term}$ = 100 – 1000

- Score 2: $TER_{bird\_short\ term} = 10 - 100$
- Score 4: $TER_{bird\_short\ term} = 1 - 10$
- Score 8: $TER_{bird\_short\ term} < 1$

3. Short-term risk to mammals
   - As for birds
4. Short-term risk to beneficial arthropods
   - Variables are:
     - Maximum application rate (g/ha)
     - Inhibition of activity (%)
   - Scoring as for hypogean soil systems
   - Final calculations for short-term risk (PRIES-1 Index):
     - PRIES-1 = [3 * Score(Bees)] + [4 * Score(Birds)] + [3 * Score(Beneficial)] + [2.5 * Score(Mammal)]
   - Weight values may be justified as follows: birds are assumed to be more endangered than mammals, due to their higher mobility; bees and other beneficial arthropods are set at the same level, with a lower weight, because scores for arthropods are taken twice in the index calculation.

Long-term scale: PRIES-2

Due to the variability of possible environmental scenarios, a PEC cannot be calculated; this index is qualitative due to the impossibility of obtaining a quantitative TER. Toxicity and exposure scores are then combined through an algorithm for the final calculation of the index. The lowest score for toxicity has been set at 0.1, instead of 0, to avoid a final score of 0 from toxicity alone.

Effect parameters:

1. Long-term effect on plants
   - A NOEL range is not indicated for plants; a rough indication of phytotoxicity instead
   - Scoring:
     - Score 0: - phytotoxic
     - Score 4: + phytotoxic
2. Long-term effect on bees
   - Based on NOEL (µg/bee)
   - Scoring:

- Score 0.1: NOEL > 100
- Score 1: NOEL = 10 – 100
- Score 2: NOEL = 1 – 10
- Score 3: NOEL = 0.1 – 1
- Score 4: NOEL < 0.1

3. Long-term effect on beneficial arthropods
   - Based on NOEL (g/ha)
   - Scoring:
     - Score 0.1: NOEL > 1000
     - Score 1: NOEL = 500 – 1000
     - Score 2: NOEL = 100 – 500
     - Score 3: NOEL = 10 – 100
     - Score 4: NOEL < 10
4. Long-term effect on birds
   - Based on NOEL (mg/kg diet)
   - Scoring as bees
5. Long-term effect on mammals
   - Based on NOEL (mg/kg diet)
   - Scoring as bees

Exposure parameters:

6. Persistence
   - Based on DT_{50} (days)
   - Scoring:
     - Score 0: DT_{50} < 10
     - Score 2: DT_{50} = 10 – 30
     - Score 3: DT_{50} = 30 – 90
     - Score 4: DT_{50} = 90 – 300
     - Score 5: DT_{50} > 300
7. Bioaccumulation
   - Based on $\log K_{ow}$
   - Scoring:

- Score 1: log K_{ow} < 2.5
- Score 1.1: log K_{ow} = 2.5 – 3.5
- Score 1.25: log K_{ow} > 3.5

8. Air affinity
   - Fugacity Level 1
   - Scoring:
     - Score 1: < 0.01 %
     - Score 1.25: 0.01 % – 5 %
     - Score 1.5: > 5 %
9. Soil affinity
   - Fugacity Level 1
   - Scoring:
     - Score 1: < 1 %
     - Score 1.25: 1 % – 20 %
     - Score 1.5: > 20 %
10. Maximum application rate
   - Scoring:
     - Score 1: < 50 g/ha
     - Score 2: 50 – 200 g/ha
     - Score 3: 200 – 1000 g/ha
     - Score 4: 1000 – 10000 g/ha
     - Score 5: > 10000 g/ha
   - Final calculations for long-term risk (PRIES-2 Index):
     - PRIES-2 = (Sum of all 5 effect scores / 5) * [Score(Air + Soil affinity) / 2] * Score(Bioaccumulation) * Score (Persistence) * Score (Max application rate)
   - Theoretically, the final value of PRIES-2 ranges between 0.1 and 187, but due to the complexity of the index, values higher than 100 are very rare.

C. FOR SURFACE WATER SYSTEMS

Short-term scale: PRISW-1

$PEC_{short\ term}$ = drift + runoff

- Assume 1m depth adjacent (20m) to the treated area

- Drift = max application rate * drift fraction
- Drift fraction assumed to be 4%
- Runoff obtained through models

1. Short-term risk to algae EC_{50}
   - $TER_{algae} = EC_{50} / PEC_{short\text{-}term}$
   - Scoring:
     - Score 0: $TER_{algae\_short\ term} > 10\ 000$
     - Score 1: $TER_{algae\_short\ term} = 1000 – 10\ 000$
     - Score 2: $TER_{algae\_short\ term} = 100 – 1000$
     - Score 4: $TER_{algae\_short\ term} = 10 – 100$
     - Score 6: $TER_{algae\_short\ term} = 2 – 10$
     - Score 8: $TER_{algae\_short\ term} < 2$
2. Short-term risk to Daphnia
   - Based on EC_{50}
   - As for algae
3. Short-term risk to fish
   - Based on LC_{50}
   - $TER_{fish} = LC_{50} / PEC_{short\text{-}term}$
   - Scoring as for algae
   - Final calculations for short-term risk (PRISW-1 Index):
     - PRISW-1 = [3 * Score(Algae)] + [4 * Score(Daphnia)] + [5.5 * Score(Fish)]

Long-term scale: PRISW-2

As for PRIES-2, a qualitative approach is used.

First, classes of water concentration (CCW, in mg/L) were established based on results from fugacity models. Then, a theoretical concentration in water (TCW, in mg/L) is calculated:

TCW = (max rate of application * CCW) / 10

- Factor of 10 assumed as a dilution factor in the receiving water body

Finally, the theoretical exposure in water (TEW, in mg/L) is calculated:

- TEW = TCW * Score(Persistence)

- DT_{50} is scored for persistence

TEW serves as a proxy for exposure in the TER calculations.

- $TER_{algae} = NOEL_{algae}/TEW$
- $TER_{daphnia} = NOEL_{daphnia}/TEW$
- $TER_{fish} = NOEL_{fish}/TEW$

TERs are scored, from 0 to 8

Final calculations for long-term risk (PRISW-2 Index):

PRISW-2 = [2 * Score(Algae) + 3 * Score(Daphnia) + 3 * Score(Fish)] * B * S

- Where B is the bioaccumulation potential (Kow)
- S is the % distribution of substance in sediment (fugacity level 1)

The role of sediments may be included in the index only as an exposure factor due to the nonavailability of toxicity data on sediment-dwelling organisms.

D. OVERALL RISK: ENVIRONMENTAL RISK INDICATOR FOR PESTICIDES (ERIP)

IF THE WHOLE ENVIRONMENT IS CONSIDERED, SCORES AND WEIGHT CHANGE. Scores are attributed for these variables:

- Air affinity, fugacity level 1
- Water affinity, fugacity level 1
- Soil affinity, fugacity level 1
- Sediment affinity, fugacity level 1
- Persistence DT_{50} (days)
- Bioaccumulation potential log K_{ow}
- Maximum rate of application

Effect on non-target organisms, epigean soil system:

- Effects on plants General phytotoxicity

- Effect on bees NOEL, LD_{50} (µg/ bee)
- Effect on beneficial arthropods (%)
- Effect on birds NOEL, LD_{50} (mg/kg)
- Effect on mammals NOEL, LD_{50} (mg/kg)
- Effect on non-target organisms, hypogean soil system:
- Effect on earthworms NOEL, LD_{50} (mg/kg)
- Effect on micro-organisms (%)

Effect on non-target organisms, surface water system

- Effect on algae 96-h NOEC, 96-h EC_{50} (mg/L)
- Effect on Daphnia 21-28 d NOEC, 48-h EC_{50} (mg/L)
- Effect on fish 14-28 d NOEC, 96-h EC_{50} (mg/L)

ERIP =

- [Average of scores for water and sediment affinity) * Average of effect scores for water) * Weighting factor +
- (Average of scores for air and soil affinity * Average of effect scores for epigean system) * Weighting factor +
- (Score(soil affinity) * Average of effect scores for hypogean system) * Weighting factor] * Score(Persistence) * Score(Bioaccumulation) * Score(Application rate)

Theoretically, the final value of ERIP is in the range 0.05 to 200; nevertheless, values higher than 100 are very rare.

Weights are assigned after the affinity*toxic effect is calculated. Then, a weighting factor of 1.5 is added to the system most at risk and 0.5 for the other two systems.

PROBABILITY OF BIRD MORTALITY

Environment Canada
Mineau (2002, 2004)

General notes:

- Field validated approach
- Toxicity as an HD_5

See Appendix B

Variables are:

- Oral LD_{50} for any bird species
- Application rate
- Octanol-water partition coefficient
- Molecular weight
- Molecular volume
- Rat oral LD_{50}
- Rat dermal LD_{50}

Using the existing LD_{50} values for many bird species, the HD_5 (Hazardous Dose at the 5% tail of the species sensitivity distribution) is derived. The HD_5 is the amount of pesticide in mg of chemical per kg of body weight estimated to lead 50% of mortality in a species more sensitive than 95% of all bird species

The physico-chemical and rat data are combined in a linear regression model to estimate the ability of a pesticide to penetrate avian skin.

The model was derived from field situations where pesticide was sprayed on foliage. To adjust to different scenarios (i.e. formulations other than spray and other methods of applications), "use pattern correction factors" are proposed:

- Pre-plant or pre-emergence surface application: 0.5
- Pre-plant or pre-emergence sub-surface application: 0.1

- Pre-plant or pre-emergence application followed by tarping: 0

- Pre-plant or pre-emergence soil application of granular, silica based: 2
- Pre-plant or pre-emergence soil application of granular, corn cob: 1
- Pre-plant or pre-emergence soil application of granular, non friable clays or cellulose: 0.2
- Pre-plant or pre-emergence soil application of granular, friable clays: 2
- Pre-plant or pre-emergence soil application of granular, not specified: 0.5
- Pre-plant or pre-emergence soil application of granular followed by tarping: 0

- Pre-plant or pre-emergence soil seed treatment, rice, millet or sorghum: 3
- Pre-plant or pre-emergence soil seed treatment, spring wheat, corn or oats: 2
- Pre-plant or pre-emergence soil seed treatment, spring barley: 1
- Pre-plant or pre-emergence soil seed treatment, winter cereals and peas: 0.4
- Pre-plant or pre-emergence soil seed treatment, rapeseed, mustard or alfalfa: 0.2
- Pre-plant or pre-emergence soil seed treatment, soybean, field beans, sugar beet, grass or potato pieces: 0.1

- Post-emergence ground foliar application: 1
- Post-emergence application of liquid on soil surface: 0.5
- Post-emergence sub-surface application: 0.1

- Aerial spray, either pre or post-emergence: 2

ECOLOGICAL RELATIVE RISK (ECORR)

Australia
Sánchez-Bayo et al. (2002)

General notes:

- Ranking is site specific

Model structure:

- Each index based on exposure/toxicity
- Separate ratio for each environmental compartment: air (A), soil (S), vegetation (V), groundwater (GW), surface water for aquatic species (Wa), surface water for terrestrial species that feed or drink from the water compartment (Wt), and sediment (SD); these can be further separated for the on-farm target, on-farm buffer, and off-farm areas
- Index scores remain separate

Elements of the model:

1. Exposure for a given compartment

   First, PECs are estimated using models that describe the fate and partitioning of residues in all environmental compartments; are based on the fugacity approach.

   Then:

   - Exposure = D * P * half-life in given compartment * BCF
     (a) Where D is the dose and P the probability of exposure in a compartment
     (b) D = PEC * Volume of matrix/Area in hectares
     (c) BCF calculated with QSAR which are based on the octanol-water partition coefficient, P is variable; equations cited in paper

   Exposure as calculated here has units of g day ha−1, but this should not be interpreted as an absolute value, i.e. it cannot be validated by field measurements but rather it is a relative value for use in the calculation of EcoRR scores.

   Exposure can also be calculated for each month of the year.

2. Toxicity for a given compartment

   $$\text{Ecotoxicity} = \frac{\text{Sum of } [(\text{toxicity geomean})_{taxon} / (S_{taxon}/N)]}{N}$$

   S_{taxon} is the number of species in one of the taxa considered for a given compartment

N is the total number of species of all taxa considered in that compartment (N is the sum of S_x)

Toxicity based on LC_{50} or LD_{50} because of their availability; data mostly from the Pesticide Manual and ECOTOX database

Chose the species and route of exposure (oral or dermal) combinations according to the compartment; can use plants and algae if data available; the same species can share several compartments

The geometric mean for a taxon is weighted by the proportion of its belonging species in each compartment (S/N); gives more weight to well represented taxa

N is a measure of biodiversity. Biodiversity data, i.e. identity and number of species present in a given ecosystem, for selected taxa in agricultural environments can be easily obtained from governmental or local group sources

When pesticide toxicity data are missing for a taxon, as often occurs with amphibians and reptiles: either the comparative assessment must exclude those taxa, or some rough estimates can be produced based on TE to other closely related taxonomic groups, i.e. toxicity data for fish can be used to extrapolate levels in amphibians, or data on mammals could be used for extrapolation in reptiles

A PESTICIDE PRIORITY LIST EVALUATION SCHEME: APPLES

Canada
Teed, R.S. (2004) for the National Guidelines and Standards Office (Environment Canada) and the CCME

General notes:

- Its intent is to establish a priority list of active ingredients (pesticides) for assessment and development of Canadian Water Quality Guidelines for the Protection of Aquatic Life.

Model structure:

- Composite index; scores are summed.

Elements of the model:

1. Presence of the active ingredient in the Canadian environment
   - Max score of 10
   - Variables are:

   a Frequency of Detection in National Monitoring/Surveillance Data (%)
   - Scoring criteria:
     - Score 5: > 80
     - Score 4: 60 - 80
     - Score 3: 40 - 60
     - Score 2: 20 - 40
     - Score 1: < 20

   o Volume sold/used per year (kg)
   - Scoring criteria:
     - Score 5: >1 000 000
     - Score 4: 100 000 – 1 000 000
     - Score 3: 10 000 – 100 000
     - Score 2: 1 000 – 10 000
     - Score 1: < 1 000

2. Environmental fate
   - Max score of 22
   - Variables are:

   a Half-life in soil (d)

- Scoring criteria:
  - Score 6: > 100
  - Score 4: 30 – 100
  - Score 2: < 30
- Octanol-water partition coefficient K_{ow} (Log P)
  - Scoring criteria:
    - Score 5: > 5
    - Score 4: 4 – 5
    - Score 3: 3 – 4
    - Score 2: 2 – 3
    - Score 1: < 2
- Soil sorption coefficient K_{oc}
  - Scoring criteria:
    - Score 5: <100
    - Score 4: 100 – 300
    - Score 3: 300 – 500
    - Score 2: 500 – 1000
    - Score 1: > 1000
- Water solubility (mg/L)
  - Scoring criteria:
    - Score 6: ≥3000
    - Score 5: 300 – 3000
    - Score 4: 30 – 300
    - Score 3: 2 – 30
    - Score 2: 0.5 – 2
    - Score 1: < 0.5

3. Aquatic toxicity
   - Max score of 16
   - Variables are:
     - Rainbow trout (Oncorhynchus mykiss) 96-hour LC50 (mg/L)
     - Water flea (Daphnia magna) 48-hour EC50 (mg/L)

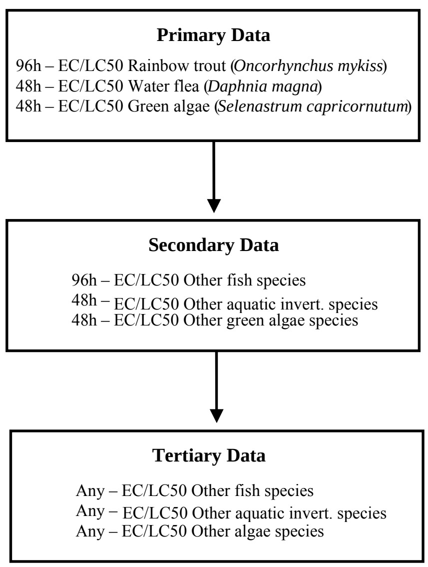
- Green algae (Selenastrum capricornutum) 48-hour EC50 (mg/L) bioassay

▪ Common risk assessment effects include mortality/survival, growth, and reproductive effects. In the case of S. capricornutum and other green algae, population effects (e.g., density)

▪ The bioassay with the most sensitive acute effect amongst the three-selected standard species is used to score toxicity. To evaluate each active ingredient, all three of the bioassay results should be available.

▪ If any of the active ingredients do not have results for the three tests, then a tiered approach is used to consider other bioassay results:

▪



▪

▪ Scoring:

- Score 1: $EC_{50}/LC_{50} > 100$
- Score 4: $EC_{50}/LC_{50} = 10 – 100$
- Score 7: $EC_{50}/LC_{50} = 1 – 10$
- Score 10: $EC_{50}/LC_{50} = 0.1 – 1$
- Score 13: $EC_{50}/LC_{50} = 0.01 – 0.1$
- Score 16: $EC_{50}/LC_{50} < 0.01$

4. Socio-political criterion

▪ Max score of 7

▪ Beyond the scope of NAESI

WWF RECOMMENDATIONS FOR PRIORITIZING RISK REDUCTION STRATEGIES BASED ON TOXICITY LOADING

by Commodity, By Region, and by Pesticide
A discussion draft by the WWF

General notes:

- Is distinct as it recommends to use commodities as a unit if analysis, instead of the active ingredient
- Specific to the Canadian agricultural environment

Model structure:

- Is an additive model (scores are summed)

Elements of the model:

Sum for all a.i. [Score(intensity of pesticide use on a given commodity) * Score(toxicity)]

Where intensity of pesticide use = pounds of a.i. per year on a given area (e.g. province)* % commodity acres treated

Toxicity score is a composite index (see below)

Source of acreage data: Canadian Census of Agriculture

Source of usage data for Canada:

- Alberta and B.C. publish pesticide sales data, but none is linked with use on specific crops.
- Every 5 years, Ontario publishes a survey of pesticide use and compiles figures on total usage of a.i. (in kg) for major commodities.
- WWF obtained breakdowns for some minor commodities, on request from the Ontario Ministry of Agriculture and Food.
- Data on herbicide usage on major cereal and oilseed crops from Manitoba and Alberta Weed Surveys conducted by the provincial and federal governments.
- Some results published from the Canola Council of Canada.
- Provincial extensions and producers groups in some cases.

Where gaps, borrowed data from regions with climatic similarities (including US data)

Toxicity score based on a modified version of Kovach's EIQ. Was modified as follows: (1) marine aquatic species were added, (2) endocrine disruption was added to the list of sub-chronic indicators, (3) the terrestrial indicator was omitted for lack of data, (4) other than the sub-chronic indicators, no toxic endpoints for human health were used.

Variables are (no details)

Acute toxicity:

LD_{50} or LC_{50}

For birds, mammals, amphibians, marine and freshwater invertebrates, marine and freshwater fish, and bees; when data permits

Chronic toxicity:

NOAEL or LOAEL (NOAEL preferred)

For birds, mammals, marine and freshwater invertebrates, marine and freshwater fish

Sub-chronic toxicity:

Carcinogenicity

Neurotoxicity

Mutagenicity

Endocrine disruption

Teratogenicity

Persistence:

Half-life in air, water, soil, and plant foliar surfaces

Soil mobility

Due to data limitations in some categories, assumptions were:

- Where amphibian data not available, freshwater fish were used
- When chronic scores for fish or invertebrates were not available, acute scores were used

- Where acute scores were not available, chronic scores were used

SCRAM: A SCORING AND RANKING SYSTEM FOR PERSISTENT, BIOACCUMULATIVE, AND TOXIC SUBSTANCES FOR THE NORTH AMERICAN GREAT LAKES

U.S.A.
Mitchell et al. 2002

General notes:

- For ranking Great Lakes contaminants

Model structure:

- Additive scoring

Elements of the model:

1. Persistence

   Variables are:

   - Media specific rate (Mackay Fugacity models)
   - Half-life for compartment

2. Bioaccumulation

   Based on BCF or log K_{ow}

3. Toxicity

   a Acute toxicity

   Lowest endpoint (EC_{50} or LC_{50}) for the most sensitive species

   b Chronic toxicity

   Lowest endpoint (NOEC or NOEL) for the most sensitive species

   c Human toxicity

   Beyond the scope of NAESI

   Data from standard Test procedures only

4. Uncertainty

   Is a reflection of data availability (of completeness of data)

   Scoring criteria are not reported (rather refers to an electronic file on the web)

   Variable scores are summed.

APPENDIX B

A PROPOSAL FOR SCORING THE MORTALITY RISK TO BIRDS FROM ACUTELY TOXIC PESTICIDES. A MODEL FOR OTHER ENVIRONMENTAL COMPONENTS?

Note: Following a round of peer review, the following is being incorporated into the PEAS (Pesticide Environmental Assessment System) system of pesticide scoring being developed in collaboration with Jennifer Curtis of Curtis Consulting (http://www.curtis-consulting.com), and Chuck Benbrook of Benbrook Consulting Services (http://www.pmac.net/). The latter has been involved for several years in the Wisconsin potato industry collaboration with the World Wildlife Fund that developed and promoted acceptance of the "Healthy Grown" brand of potatoes certified by the organization 'Protected Harvest' (http://protectedharvest.org/). PEAS is being developed to establish a scientifically sound basis for pesticide risk standards that may eventually be incorporated into an ecolabel.

Deriving the likelihood of avian mortality:

- The proposed risk assessment for birds is unique among the other components. Whereas it is customary to have some form of TER (Toxicity/Exposure Ratio) or RQ (Risk Quotient) at the core of most indicators, we have relied instead on the logistic models developed in the course of previous analyses of avian field studies (Mineau 2002)[1] in order to derive a likelihood that a given pesticide application will result in observable avian mortality. So, in essence, the avian index has already been validated against real field outcomes – unlike most calculated ratios of exposure and toxicity.

- The process can be summarized as follows: As a first step, a measure of acute pesticide toxicity for birds ranging from 20 to 1,000 grams (a weight range that covers most bird species found dead in farm fields) is obtained by applying species sensitivity distribution techniques (Mineau et al. 2001)[2.] A value called the HD5 ('Hazardous Dose at the 5% tail of the species distribution') is derived. The HD5 is the amount of pesticide in mg of chemical per kg of body weight estimated to lead to 50% mortality in a species more sensitive than 95% of all bird species, calculated with a 50 percent probability of over- or

[1] *Mineau, P. (2002). Estimating the probability of bird mortality from pesticide sprays on the basis of the field study record. Environmental Toxicology and Chemistry 24(7):1497-1506.*
[2] *Mineau, P., A. Baril, B.T. Collins , J. Duffe, G. Joerman, R. Luttik (2001). Reference values for comparing the acute toxicity of pesticides to birds. Reviews of Environmental Contamination and Toxicology 170:13-74*

underestimation. The HD5 can be calculated mathematically where several toxicity values exist, or extrapolation factors can be applied to single (or even multiple combinations of species-specific toxicity values – see Table 1 in Mineau et al. 2001).

- A probability of kill is then derived from a model that uses logistic multiple regression with the finding of bird carcasses in fields as the endpoint of interest. Note that this index does not incorporate other toxic effects on birds, or indirect effects. (The latter would probably best be captured in a terrestrial invertebrate index.) Aside from the HD5 values, the model makes use of application rate, as well as physico-chemical constants such as octanol-water partition coefficient, molecular weight and size as well as the ratio of rat oral to dermal data, if available. The physicochemical and rat data are combined in a linear regression model to estimate the ability of pesticides to penetrate avian skin. This ability has been found to significantly affect field outcome. Based on extensive experience with the models, it has been found that the risk of bird mortality is negligible for any product with an HD5 greater than 100. In those cases, the models are not run and the probability that the pesticide in question will give rise to visible mortality is set at 0%. Most herbicides and fungicides as well as some insecticide families (e.g. most synthetic pyrethroids) fall into this negligible risk category. Independent validation of the model for a sample of studies in field crops indicate that better than 81% of studies were correctly classified – as to whether they gave rise to mortality or not.

- One recognized weakness of the approach is that the empirical models relating mortality to HD5 and to the other independent variables were derived entirely from foliar applications of pesticides. Adjustment factors based on the best available expert opinion are needed to integrate the avian exposure-related consequences of alternative pesticide formulations (e.g. granular, seed treatment), methods of application (ground sprayer, air-blast, aerial), and timing of application. A draft version of the “Use Pattern Adjustment Factors” (UPAFs) was circulated to a panel of experts in avian pesticide impact and triggered extensive dialogue. This updated description of the model and provisional UPAFs has benefited in several ways from this dialogue. However, the final choices and compromises that needed to be made are those of the author. The UPAFs proposed here are attempts to express the risk associated with a given type of pesticide application relative to the risk posed by a foliar spray. For example, an adjustment factor of 2 means that the risk of avian mortality from a given type of application is roughly twice what it would be if the same a.i. was foliar applied by ground rig at the same rate per hectare.

Derivation of UPAFs:

Proposed adjustment factors are given in tables 1-3. A short justification is provided for the proposed factors.

APPENDIX B: Table 1: Proposed avian "Use Pattern Adjustment Factors"

Pre-Plant or Pre-Emergence				Post-Emergence		Either
Soil Applied: Liquid	**Soil Applied: Granular**	**Soil Applied: Unspecified**	**Seed Treatment**	**Ground Foliar Applied**	**Soil Applied: Liquid**	**Aerial Application**
0.5 (surface) 0.1 (sub-surface) 0 (application followed by tarping)	See below	0.5	See below	1	0.5 (surface) 0.1 (sub-surface)	1

Soil applied vs. foliar applications

Applications to bare soil should reduce potential exposure to birds relative to foliar applications. Exposure is still possible through contamination of drinking puddles as well as contamination of ground dwelling arthropods. However, the extent of dermal exposure should be less than in a foliar situation. Also, contamination of most phytophagous insects, a primary bird food source, is avoided. Subsurface application should reduce the potential for exposure even more considerably. Risks of surface and subsurface liquid applications have been set at half and one tenth of the risk of a foliar application respectively.

Granular applications

In the case of avian risks, some granular formulations are very attractive to birds and pose far more risk than a liquid application. Similarly, some seed treatments represent a high source of exposure.

Silica based granular pesticides are widely acknowledged to be the most attractive to birds and therefore the most dangerous (Fischer and Best 1995)[3]. In a review of carbofuran field studies conducted by the manufacturer, Mineau (1993)[4] reported raw uncorrected bird carcass counts of 0.43 and 0.53 carcasses per ha for foliar applications of 1.1 kg ai/ha carbofuran to mature corn. This compares to uncorrected kill rates of 0.23 and 1.5 carcasses/ha following pre-plant incorporated band applications of 1.5 kg ai/ha of the granular silica formulation in corn. Other studies, however, have documented higher kill rates – for example one granular study in Utah with uncorrected kill rates of 8.9 carcasses per ha in corn – albeit at a higher rate of application. This is clearly a very uncertain comparison but, looking at median kill rates in the case of liquid and granular products suggests that the risk of a silica based granule is at least twice as high as that of an equivalent foliar application. It may be much higher still. However, this particular UPAF will have minimal effect overall because few granular products are formulated on silica – fortunately.

Corncob granular formulations on the other hand have been shown to be less attractive as grit but their attractiveness increases as the availability of food diminishes. The risk from corncob granular formulations is (arbitrarily) set equal to that of a foliar spray. This may be an underestimate of the risk, especially with the more concentrated granular products (e.g. 15G or 15% by weight) but, more information will be needed to assess this. Reviewers have suggested that the risk of granulars be assessed on their own based on granule type, toxicity, concentration and some of the other characteristics that may influence attractiveness in the field. This is clearly what we should strive to do. Unfortunately, there has not been any validation of such a scheme to date.

[3] *Fischer, D.L. and L.B. Best. (1995). Avian consumption of blank pesticide granules applied at planting to Iowa cornfields. Environmental Toxicology and Chemistry. 14(9): 1543-1549.*

[4] *Mineau, P. 1993. The hazard of carbofuran to birds and other vertebrate wildlife. Technical Report Series. No. 177. Environment Canada, Canadian Wildlife Service, Ottawa. XXii+96 pp.*

Clay carriers, whether bentonite or montmorillonite clay as well as cellulose products, are thought to be the least attractive to birds because they are neither satisfactory as food or as grit. Limited field studies (e.g. Knapton and Mineau 1995)5 have failed to confirm a high risk from clay formulations of fairly toxic granular products. However, choice experiments have shown that birds will sample these granules also and kills have been reported. The risk of puddling and contamination of both drinking water and ground dwelling arthropods is possible although lower than for a surface ground application because some incorporation is often the norm with granular products. Also, contamination of field edges is probably less than for a foliar surface spray. Granule types, which are very friable and break down quickly (bentonite and gypsum), are thought to be the safest. The risk for resilient heat-treated clay granules is therefore set half way between that of a surface and subsurface liquid application. The risk from the more friable granule types is set at half that – or 1/10th the risk form a corncob formulation.

Avian UPAFs for granular applications are given in Table 2. At this point, we have chosen not to let the granular application equipment affect the adjustment factors. Even though side dressing of granules post-emergence and even banded applications leave many more surface granules than in furrow applications (for example), some studies (e.g. Fischer and Best, op. cit., Stafford and Best 19976) suggest that the risk is less affected by the number of surface granules (there always being an excess) than by the granule type and toxic loading per granule. On the other hand, granules that are applied and then immediately tarped (such as fumigants) should not carry much of a risk of ingestion by birds.

APPENDIX B: Table 2. Avian Exposure Use Pattern Adjustment Factors for granular applications

Silica granules	**Corn cob (organic) granules**	**Heat treated montmorillonite and other non friable clays, cellulose**	**Friable granule bases: bentonite and gypsum**	**Tarping follows granular application**
2.0	1.0	0.2	0.1	0

[5] *Knapton, R.W. and P. Mineau.1995. Effects of granular formulations of terbufos and fonofos applied to cornfields on mortality and reproductive success of songbirds. Ecotoxicology 4(2):137-152.*
[6] *Stafford, T.R. and B. Best. (1997). Effects of granular pesticide formulations and soil moisture on avian exposure.*

Seed treatments

Treated seeds have historically represented an important source of exposure for birds and toxic seed dressings do cause bird mortality (Greig-Smith 1987)[7]. However, it is widely acknowledged that not all seed types are equally attractive to birds. The attractiveness of different agricultural seeds has been found to vary approximately 30 fold in recent British research (Prosser 2001)[8]. In order to establish preferences (and hence risk) in the case of North American bird species, we used a weighted average of seed attractiveness based on the number of bird species documented to make use of the seed type as well as the proportion of the various species' diet this represents. The information was summarized from Martin et al. (1951)[9] and is based on extensive bird collections carried out by the USDA at the turn of the 20th century. This analysis revealed an approximately 20-fold difference between the most and least sought after seed types. Unfortunately, we were unable to correct for the relative surface area cultivated. Also, not all seed types were included and we used best expert judgment as well as the British data previously cited to establish the following exposure adjustment factors. Furthermore, the data summarized in Martin et al. do not differentiate between seed taken directly from the plant and seed taken from the ground at seeding time. This is potentially a serious source of uncertainty viz. the relative risk of seed treatment chemicals. Finally, a number of factors cannot be taken into account – such as the specifics of the seed coating, impacts of polymers and stabilizers, and amount of pelletization.

Tentative factors are provided in Table 3: A seed type of average attractiveness (barley) was set at 1 – i.e. with an inherent risk as high as that of a foliar application. Again, this is a rather arbitrary placement subject to changing if new data come to light. Other seed types were grouped and the groups ordered relative to barley (Table 3).

[7] *Greig-Smith, P. W. Hazards to wildlife from pesticide seed treatments. 39, 127-134. 1987. British Crop Protection Coucil. BCPC Monograph.*
[8] *Prosser, P. (2001). Project PN0907: Potential exposure of birds to treated seed. Final milestone report (Revised edition: 5 March 2001) Central Science Laboratory, December 1999, Unpublished.*
[9] *Martin,A.C.; Zim,H.S.; Nelson,A.L. 1951. American Wildlife & Plants: A Guide to Wildlife Food Habits: The Use of Trees, Shrubs, Weeds, and Herbs by Birds and Mammals of the United States. Dover Publications, Inc.*

APPENDIX B: Table 3: Avian Exposure Use Pattern Adjustment Factors for Seed Treatments

Rice Millet Sorghum	Spring wheat Corn (maize) Oats	Spring barley	Winter cereals Peas	Rapeseed, mustard Alfalfa	Soybean, field beans Sugar beet Grass Potato pieces (?)
3.0	2.0	1.0	0.4	0.2	0.1

Aerial application

Mineau (2002) found broad overlap between the risks of aerial and ground applications of field crops, although he was able to document higher risks from forest spraying where smaller droplet sizes are the norm. Non-crop field edge habitat is much more likely to be contaminated following aerial application than with an equivalent ground application. Therefore, we had initially proposed to use a correction factor above 1 to account for the more widespread contamination from an aerial application – and higher probability of exposure. However, this was criticized by some reviewers because we were not taking into account the lower deposit per surface area that results from aerial application. Given that we were not able to separate aerial from ground applications (Mineau op. cit.), it may be prudent at this time to assume that the two opposing risk factors – more extensive contamination vs. reduced deposits – cancel each other out and not differentiate between ground and aerial application.

Crop type

Not all crops are visited by birds with the same frequency and it is tempting to apply some factor to represent the extent of avian use (and hence increased risk) associated with some crop types. However, as argued by Mineau (2002) few crops can be said to be completely devoid of bird activity. Also, based on the work of Best and unpublished industry data, Solomon and colleagues (2001)10 estimated that the proportional use of field centres relative to non-crop edge habitat ranged from 0.5% to 86% depending on the species in mid-west corn fields. The median species had 88% of its activity recorded in the field edges. This argues against applying crop-specific correction factors given that the quality of the field edge may be more important than the crop type in defining the bird species and numbers at risk.

[10] *Solomon et al. (2001). Chlorpyrifos: Ecotoxicological risk assessment for birds and mammals in corn agroecosystems. Hum. Ecol. Risk Assess. 7(3):497-632.*

Scaling of the risk

Examples of possible outcomes with a simple multiplicative effect are given in table 4. The most hazardous condition (the seeding of rice seed) and a kill probability of 1 (as determined from the models described in Mineau 2002) yield a worst case Avian Risk Index of 3.0. At the opposite end of the spectrum, risk from a highly toxic product is reduced to naught through measures that exclude bird exposure, such as tarping.

APPENDIX B: Table 4. Examples of possible Avian Risk Scores when the likelihood of mortality (from 0 to 1) is combined in a scalar fashion with the Use Pattern Adjustment Factors obtained from expert opinion.

Risk of detectable avian mortality - model output from Mineau 2002 on a scale of 0 to 1.		Examples of application types and applicable adjustment factors					
		Application of rice seed	**Seeding of corn or use of silica granules**	**Foliar treatment**	**Heat-treated clay granules**	**Sub-surface liquid**	**Soil surface application and tarping**
	Use pattern adjustment factor →	**3**	**2**	**1**	**0.2**	**0.1**	**0.5*0=0**
1		3	2	1	0.2	0.1	0
0.8		2.4	1.6	0.8	0.16	0.08	0
0.5		1.5	1	0.5	0.1	0.05	0
0.2		0.6	0.4	0.2	0.04	0.02	0
0		0	0	0	0	0	0

Although it may be argued that a resulting score of 3 represents a level of risk which is higher than, say, a score of 1.5, both applications are predicted to result in avian mortality on every treated field. By virtue of the higher score, it may be logical to assume that there should be more mortality following the application with a score of 3 than the one with a score of 1.5. However, the original risk calculation of Mineau (2002) did not distinguish on the basis of extent of mortality but, rather, on the probability that some mortality would be observed. It is recognized that extent of mortality depends first and foremost on the number of birds present and 'available to be killed'. Therefore, it does not seem prudent to attribute much significance to the difference between scores of 1 and those that are over 1. we therefore propose that the risk index should plateau at 1 and hence continue to represent the likelihood of mortality between 0 and 100% (Table 5).

One problem with the way in which use pattern adjustment factors are applied may be at the lower risk levels. If a product does not carry any risk of mortality when applied as a foliar spray (initial probability of mortality of 0), does it follow that application of a silica granular or seed treatment with the same a.i. carries no more risk as implied by the bottom rows of tables 4 and 5. Logically, this is a contradiction given that we have already established the risk of those formulation types to be inherently greater than the equivalent foliar application. Because we are currently unable to deal with this problem, this is something that will need to be considered as we gain more experience with the avian risk index and are able to assess the 'reasonableness' of the predictions.

APPENDIX B: Table 5. Examples of possible Avian Risk Scores from Table 4 adjusted to reflect a risk 'plateau' of 1.

Risk of detectable avian mortality - model output from Mineau 2002 on a scale of 0 to 1.		Examples of application types and applicable adjustment factors					
		Application of rice seed	**Seeding of corn or use of silica granules**	**Foliar treatment**	**Heat-treated clay granules**	**Sub-surface liquid**	**Soil surface application and tarping**
	Use pattern adjustment factor →	**3**	**2**	**1**	**0.2**	**0.1**	**0.5*0=0**
1		1	1	1	0.2	0.1	0
0.8		1	1	0.8	0.16	0.08	0
0.5		1	1	0.5	0.1	0.05	0
0.2		0.6	0.4	0.2	0.04	0.02	0
0		0	0	0	0	0	0

Aggregation

Because the final risk indices are probabilities, it is possible to aggregate pesticide impacts temporally or spatially in the form of kill-hectares. For example, a pesticide treatment carrying a risk index of 0.6 and applied to 1000 hectares has a aggregate impact of 600 'kill-hectares'. Different pesticide treatments within a field can be aggregated independently of each other even if the resulting number of 'kill-hectares' exceeds the number of planted hectares. This is because treatments of high avian toxicity are usually spaced within the growing season and their respective impacts are probably additive. By aggregating kill-hectares over time or within a defined area, one can look at trends to see whether the risk of mortality is improving or worsening for birds in our farmland. There is growing evidence that the concept of kill-hectares has merit because it does help explain bird population declines observed in farmland (Mineau, unpublished analysis).

Establishment of Standards

Standards can only be set once all interveners have agreed on an unacceptable level of avian mortality in farm fields. Despite some of the assumptions that had to be made in arriving at a measure of acute risk to birds, the index provided is a measure of true validated risk as opposed to a measure of hazard. Ideally, and in conformity with the Migratory Birds Convention Act, the risk of avian mortality following application of a pesticide should be negligible. Realistically, it may be some time before those products causing bird mortality are removed from the market[11]. Therefore, the setting of a standard will require that some probability of kill be judged to be too high.

[11] For a discussion of this issue, see: *Mineau, P. 2004. Birds and pesticides: Are pesticide regulatory decisions consistent with the protection afforded migratory bird species under the Migratory Bird Treaty Act ? The William and Mary Environmental Law and Policy Review 28(2): 313-338.* Reprints are available from the author at Pierre.mineau@ec.gc.ca.