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Georeferenced Probabilistic Risk Assessment of Pesticides –

Further Advances in Assessing the Risk to Aquatic Ecosystems by Spray Drift from Permanent Crops

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Georeferenced Probabilistic Risk Assessment of Pesticides –

Further Advances in Assessing the Risk to Aquatic Ecosystems by Spray Drift from Permanent Crops

by

**Kubiak R. & Hommen U., Bach M., Classen S.,
Gergs A., Golla B., Guerniche D., Klein M., Krumpe J.,
Preuss TG., Ratte HAT., Roß-Nickol M., Schäfers C.,
Strauss T., Toschki A., Trapp M.**

Project coordination

Prof. Dr. Roland Kubiak
Institute of Agroecology, RLP Agroscience, Neustadt a.d. Weinstraße

Dr. Udo Hommen
Fraunhofer Institute for Molecular Biology and Apllied Ecology, Schmallenberg

On behalf of the Federal Environment Agency (Germany)

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info@umweltbundesamt.de
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Vorwort

Das Umweltbundesamt unterstützt die Entwicklung von landschaftsbezogenen probabilistischen Methoden in der Umweltrisikobewertung von Pestiziden bereits seit 2003[1] mit eigenen Beiträgen einschließlich der Vergabe von Forschungsvorhaben. Inzwischen ist die Anwendung landschaftsbezogener probabilistischer Methoden in der Umweltrisikobewertung von Pestiziden nicht mehr nur Diskussionsgegenstand der Wissenschaftsgemeinde, sondern hat auch die Ebene der Risikoregulierung sowohl auf nationaler Ebene als auch auf EU-Ebene erreicht. Das vorliegende Forschungsvorhaben wurde vom Umweltbundesamt in Auftrag gegeben, um basierend auf bereits bestehenden Vorarbeiten auf nationaler Ebene[2] ein fachlich fundiertes Gesamtkonzept für die Anwendung von landschaftsbezogenen probabilistischen Methoden in der Risikobewertung zu erarbeiten und die damit verbundenen technischen und prozeduralen Anforderungen für eine Implementierung bei der Produktzulassung aufzuzeigen. Die Arbeiten fokussierten auf Einträge von Pestiziden in Gewässern über die Pfade ´Abdrift´ und ´Verflüchtigung und Deposition´ aus der Anwendung in Dauerkulturen. Eine wesentliche Anforderung des Umweltbundesamtes an die Autoren der Studie war, dass mit dem neuen Bewertungsansatz weiterhin ein hohes Schutzniveau für alle Gewässerorganismen gewährleistet sein sollte.

Die derzeitige Expositionsabschätzung für Einträge von Pestiziden in Gewässer beruht auf der Betrachtung eines Modellgewässers – ein sog. ´Realistic worst case´- Gewässer der Agrarlandschaft. Für die Faktoren, welche die zu erwartende Pestizidbelastung im Gewässer bestimmen, werden eine Reihe deterministischer Annahmen für die Eigenschaften und Lage des Modellgewässers zu Grunde gelegt, wie z.B. eine angenommene Standardtiefe von 30 cm eines stehenden Gewässers und die unmittelbare Nähe zum Feldrand. Dieser sog. szenarienbasierte Bewertungsansatz ermöglicht eine protektive Risikobewertung, erlaubt jedoch keine Einschätzung des zeitlich-räumlichen Auftretens von Expositionssituationen in der Landschaft. Wo keine räumlich explizite Information zu Standortfaktoren mit relevantem Einfluss auf die Exposition und das Risiko in einem Gewässer vorliegen, kann diese auch nicht im Rahmen des Risikomanagements berücksichtigt werden. Die Anwendung georeferenzierter probabilistischer Methoden in der Risikobewertung erlaubt hingegen eine standortbezogene sowie quantitative Charakterisierung der Exposition und Risiken von Pestizidrückständen in Gewässern. Sie eröffnen damit auch die Möglichkeit für ein räumlich differenziertes Risikomanagement. Damit verbunden ist die Erwartung, dass auf dieser Basis abgeleitete Umweltauflagen für Anwender durch den direkten Standortbezug nachvollziehbarer und somit eventuell akzeptabler sein können. Gleichzeitig ermöglicht der Ansatz auch eine Fokussierung der Anstrengungen landschaftsbezogener Risikomanagementmaßnahmen auf stärker gefährdete Gewässerbereiche.

Die Ergebnisse des Forschungsvorhabens zeigen die grundsätzliche Umsetzbarkeit einer landschaftsbezogenen Expositions- und Risikoabschätzung für Gewässereinträge von Pestiziden über die o.g. Pfade auf. Hierbei wird sehr deutlich, dass sowohl aus Sicht der methodischen als auch technischen Anforderungen die Implementierung des entwickelten GeoRisk-Ansatzes auf bundesweiter Ebene zum derzeitigen

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1 Workshop Proceedings UBA/IVA/BVL – Workshop on Probabilistic Assessment Methods for Risk Analysis in the Framework of Plant Protection Products Authorization Berlin, 25 – 28 November 2003

2 Schulz et al. (2010): „Umsetzung der georeferenzierten probabilistischen Risikobewertung in den Vollzug des PflSchG – Pilotphase – Dauerkulturen“, UFOPLAN-Vorhaben (FKZ: 206 63 402)

Entwicklungsstand nicht ohne Weiteres möglich wäre. So zeigen die Autoren die Implementierung eines dynamischen Gewässermodells für Fließgewässer als eine Voraussetzung für eine realitätsnahe Expositionsabschätzung auf Landschaftsebene auf. Im Projekt war es aufgrund der hohen mathematischen Komplexität der dynamischen Modellierung jedoch nicht möglich, die komplette Netzwerkstruktur bei der Modellierung der Belastungen in den jeweiligen Segmenten abzubilden. Um eine ausreichende Sicherheit der Risikoprognose zu gewährleisten, ergibt sich daher ein zusätzlicher Bedarf für die Weiterentwicklung des Ansatzes. Weiterhin ist eine Umsetzung auch wegen hoher Anforderungen des dynamischen Ansatzes an die Verfügbarkeit hochauflösender GIS-Daten zu standörtlichen Expositionsfaktoren wie z.B. der Lagebeziehung zwischen Anwendungsflächen und Gewässern als auch der Hydrodynamik der jeweiligen Abschnitte die Umsetzbarkeit derzeitig nur auf kleinerer regionaler Ebene möglich. Die derzeitige Zulassung von Pflanzenschutzmittel in einem zonalen Bewertungsverfahren ist mit dem Anspruch auf eine Harmonisierung der Bewertungsgrundlagen innerhalb einer Zone verbunden. Die Anwendung landschaftsbasierter Ansätze für die Verfeinerung der Risikobewertung zu Pflanzenschutzmittelprodukten ist daher auch mit einer Reihe von Fragen zur regulatorischen Umsetzbarkeit verbunden. Wie kann z. B. ein Mitgliedsstaat die Produktzulassung eines anderen Mitgliedsstaates in seiner Zone anerkennen, wenn ein vertretbares Risiko nur auf Basis einer standortspezifischen Verfeinerung der Bewertung und des Management dieser Risiken gezeigt werden kann?

Allerdings stellt die Studie aus regulatorischer Sicht einen wichtigen Beitrag zur aktuellen wissenschaftlichen Diskussion zur Weiterentwicklung der Umweltrisikobewertung dar, insbesondere dadurch, dass eine ökologisch relevante raumzeitliche Ebene in der Exposition von Gewässern und der daraus resultierender Auswirkungen auf aquatische Organismen und folgende Wiedererholungsprozesse betrachten wird. Eine wesentliche Erkenntnis der Autoren ist insbesondere, dass die Identifikation sog. „Hotspots" (Gewässerstrecken mit einem ökologisch kritischen Eintragsrisiko) und deren Management durch lokale Risikominderungsmaßnahmen als unabdingbarer Bestandteil eines landschaftsbezogenen probabilistischen Ansatzes in der Risikobewertung verstanden werden muss, um weiterhin ein hohes Schutzniveau für die für aquatische Ökosysteme sicherstellen zu können. Mehr Realitätsnähe in der Risikobetrachtung durch Berücksichtigung des landschaftlichen Kontextes ermöglicht eine Verringerung derzeit erteilter Auflagen für weniger gefährdete Abschnitte von Gewässern nur bei einem gleichzeitigen lokalen Risikomanagement von hoch gefährdeten Gewässerabschnitten, in denen kritischen Umweltkonzentration aggregiert auftreten.

Foreword

The German UBA has already supported the development of landscape-level and probabilistic approaches in the environmental risk assessment with own scientific contributions including R & D projects since 2003. Since that time the use of landscape-level and probabilistic approaches in risk assessment is not any more defined to the discussion in the scientific community but has also reached the world of the pesticide risk regulators at national and at EU level. The current R & D project has commissioned by the German UBA in order to establish a scientifically sound overall approach for the implementation of a geo-referenced probabilistic approach in risk assessment at national level. Furthermore the authors illustrate the technical and procedural requirements which would be linked to an implementation of such an approach for the national product authorization. The work focused on pesticide entries into water bodies via ´drift´ and ´volatilization and deposition´ due to their application in permanent crops. The essential requirement for the study authors was to develop an approach that would continue to ensure a high protection level for aquatic organisms.

The current exposure assessment for pesticide entries into water bodies considers a water body model which represents a realistic worst case for water bodies in the agricultural landscape. For the factors determining the pesticide exposure in the model a variety of deterministic assumptions are made such as that it is a stagnant water body of a default depth of 30 cm which is directly situated in the edge of the field. Although this scenario-based approach enables a protective risk assessment it does not allow for an estimation of the spatiotemporal occurrence of this exposure situation in the landscape. Spatially explicit information on location factors with a relevant impact on exposure and risk in a water body are missing and can therefore not be considered in the risk management. In contrast the use of geo-referenced probabilistic methods in risk assessment allows for a site-specific as well as quantitative characterization of the pesticide exposure and linked risks in water bodies. Consequently such approaches also establish the possibility for a spatially differentiated risk management. This again leads to the expectation that such risk mitigation obligations due to their direct reference to the real local conditions might be more plausible and therefore acceptable for farmers. Furthermore they also give possibility to focus efforts in permanent local risk mitigation measures on those water body sections with a high risk.

The project results demonstrate the general feasibility of the implementation of a landscape-level probabilistic risk assessment for pesticide entries in water bodies via the above mentioned paths. However it is obvious that the implementation of the suggested ´GeoRisk´ approach at federal level at the current state of development is not possible considering both the methodological as well as technical requirements of the approach. Thus, the authors point out that the realization of a dynamic fate model for streaming water bodies is an essential demand for a realistic exposure assessment at landscape level. However, within the project to it was still not possible to reflect the full water network structure in the modeling of the pesticide contamination in the respective sections of the water network due to the high mathematical complexity of the dynamic modeling approach. Therefore a need for further development of the approach is indicated in order to ensure a sufficiently certain risk prognosis. Furthermore due to the high demands regarding the availability of high resolution GIS data on the site-specific exposure determining factors such as the spatial relation between application areas and adjacent water bodies as well as the hydro-dynamical characteristics of a specific network section an implementation seems currently only possible on a smaller regional scale. The current authorization procedure on the base of a zonal peer review of the risk assessment has the aim to ensure a high degree of harmonization in the principles of risk assessment. Therefore the use of landscape-level approaches in risk assessments is not only linked to scientific but also a variety of regulatory questions. For example the question whether a member state can recognize the product authorization of another member state in its zone if an acceptable risk for that product is only demonstrated on the base of a landscape level and therefore very site specific approach in risk assessment and management.

However, from a regulatory point of view the study represents an important contribution to the current scientific discussion of further advancements in environmental risk assessment especially by considering an ecologically relevant spatiotemporal scale of pesticide exposure in water bodies and resulting impacts on aquatic organisms and subsequent recovery processes. One of the essential scientific finding of the authors in particular is that the identification of so-called ´hot spots´ (water body section with an ecologically critical pesticide load risk) and their subsequent management by site-specific risk mitigation measures has to be understood as an indispensable part of a geo-referenced probabilistic risk assessment and management approach in order to continue to ensure a high protection level for aquatic ecosystems. More realism in environmental risk assessment due to consideration of the landscape level enables on the one hand the reduction of recent risk mitigation measures in less vulnerable water body sections but on the other hand consequently requires to address those water body sections with local risk mitigation measures which are at high risk due to the clustering of critical environmental concentrations of pesticides.

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15. Supplementary Notes
Supplementary stand-alone documents are Appendices A to D and a software application:
Appendix A: GeoRisk Framework document; Appendix B: GeoRisk Technical Manual for the GIS tool;
Appendix C: GeoRisk Workshop Report; Appendix D: Manuskript Recovery Review
Software application: geoRISK WEB-System

16. Abstract

Within the GeoRisk project a new approach for the aquatic risk assessment of plant protection products (PPPs) used in permanent crops (vine, orchards and hops) in Germany is further developed and evaluated. The aim is to establish a more realistic risk assessment by the use of geodata and probabilistic methods. As a consequence, the derivation of spatially specific and therefore more appropriate risk management measure is possible nevertheless ensuring the protection of non- target aquatic species.

The GeoRisk approach is based on the following key elements:

- Geodata based probabilistic calculation of drift and volatization / deposition entries of plant protection products into edge of field water bodies and of initial concentrations in lentic water bodies
- Consideration of dispersion and transport of PPPs in running waters by a dynamic exposure model
- Identification of the ecologically critical aggregation of water body segments with high risks('hotspots') considering tolerable effect levels for populations of aquatic species
- Implementation of a spatially differentiated risk management in the identified hotspots
- Confirmation of possibility for authorization of products based on the risk of new product related hotspots.

One of the main outcomes of the project is the recommendation to use the more realistic dynamic exposure model instead of the formerly favoured static model. However, due to the complexity of this newly developed model and missing data for several input parameters it has not been possible within this project to reach a stage for nationwide implementation yet. Suggestions for the steps required to implement the approach including the necessary hotspot management in all permanent crop areas in Germany were elaborated.

The GIS-based tool 'geoRISK-WEB' including data base and data management tool was developed to identify hotspots and risk management segments.

17. Keywords
aquatic organisms, drift, dynamic exposure model, ecological traits, exposure pattern, geodatabase, georeferenced, GeoRisk, geoRISK-WEB, GIS modeling, hotspot, life-history, plant protection product, pesticide, products, probabilistic, permanent crops, recovery, re-colonization, risk management segments, risk assessment, risk management, spatially explicit

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Claßen, Silke	Preuss, Thomas	Toschki, Andreas
Golla, Burkhard	Ratte, Toni	Trapp, Matthias

6. Durchführende Institution (Name, Anschrift)

Konsortium unter der Leitung von:
RLP AgroScience GmbH
Institut für Agrarökologie
Breitenweg 71
67435 Neustadt

7. Fördernde Institution (Name, Anschrift)

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15. Zusätzliche Angaben
Als eigenständige Dokumente ergänzen die Anhänge A bis D sowie ein Software-Tool den Projektbericht:
Anhang A: GeoRisk Framework document; Anhang B: GeoRisk Technical Manual for the GIS tool;
Anhang C: GeoRisk Workshop Report; Anhang D: Manuskript Recovery Review
Software GIS Tool: geoRISK WEB-System

16. Kurzfassung
Im GeoRisk-Projekt wurde für Obst-, Wein- und Hopfenkulturen in Deutschland ein neuer Ansatz der Risikobewertung von Pflanzenschutzmitteln auf der Basis von Geodaten und probabilistischen Verfahren weiter entwickelt und bewertet. Der Ansatz ermöglicht die Ableitung stärker lokalitätsbezogener und somit geeigneterer Risikomanagementmaßnahmen ohne den Schutz der aquatischen Populationen zu verringern.
Der GeoRisk-Ansatz beruht auf den folgenden fünf Elementen:

- Georeferenzierte probabilistische Berechnung von Gewässereinträgen über Drift und Verflüchtigung/Deposition in der Nähe von Applikationsflächen sowie der sich ergebenden Initialkonzentrationen in stehenden Gewässern
- Berücksichtigung von Transport und Verdünnung in Fließgewässern durch ein dynamisches Expositionsmodell
- Identifizierung ökologisch kritischer Häufung von Gewässerabschnitten mit hohem Risiko von Effekten (‚Hotspots') auf der Basis tolerierbarer Effekte aquatischer Populationen
- Implementierung eines räumlich differenzierten Risikomanagements in den Hotspots
- Bestätigung der Zulassungsfähigkeit von Produkten, wenn durch deren Anwendung keine neuen Hotspots entstehen.

Ein Ergebnis des Projektes ist die Empfehlung eines dynamischen Expositionsmodells anstatt des bisher geplanten statischen Modells. Für eine flächendeckende Umsetzung in Deutschland liegen jedoch noch nicht ausreichend Eingangs-Geodaten vor. Vorschläge für die notwendigen Schritte zur Einführung des Ansatzes in allen Raumkulturen in Deutschland inklusive der Implementierung des notwendigen Hotspotmanagements wurden erarbeitet. Die Software-Anwendung ‚geoRISK-WEB' inklusive Geodatenbank und Daten-Management-Tool ermittelt Hotspots' bzw.die Anzahl von Risikomanagementsegmenten, als Kriterium für die Zulassung von PSM-Anwendungen.

17. Schlagwörter
Abdrift, aquatische Organismen, Anwendungsbestimmungen, Dauerkulturen, dynamisches Expositionsmodell, Expositionsmuster, Geodatenbank, georeferenziert, GeoRisk
geoRISK-WEB, GIS Modellierung, Hotspot, Lebenzyklus, ökologische Eigenschaften, Pestizid, Pflanzenschutzmittel, Probabilistik, Risikomanagementsegmente, Risikobewertung, Risikomanagement, Wiedererholung, Wiederbesiedlung

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Content – Main chapters

Appendices

Detailed content

Appendices

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Abbreviations

ATKIS	Amtliches Topographisch-Kartographisches Informationssystem
BBA	Biologische _Bundesanstalt
BDLM	Basic Digital Landscape Model (Basis Digitales Landschaftsmodell)
BKG	Bundesamt für Kartographie und Geodäsie
BVL	Bundesamt für Verbraucherschutz und Lebensmittelsicherheit
EC_x	Concentration resulting in x % effect
EDB	Exposure database
EU	European Union
GIS	Geographic Information System
HC_5	Hazardous Concenration for 5 % of the species, 5^{th} centile of a SSD
HR	High resolution (related to geodata)
IME	Fraunhofer Institute for Molecular Biology and Applied Ecology
IVA:	Industrieverband Agrar
JKI	Julius Kühn Institute
LCA	Land Consolidation Authority (Local authorities responsible for agricultural advice, for land management and for local environmental protection), different authority structures in the different German states
LDB	Landscape database
MC	Monte-Carlo simulation
MS	Management segment
NDB	Network database
NOEAEC	No Observerd Ecologically Adverse Effect Concentration
NOEC	No Observed Effect Concentration
PEC	Predicted Environmental Concentration
PECini	initial PEC
$PEC_{TWA(1h)}$	Maximum time weighted average PEC over 1 h
PPP	Plant Protection Product
RAC	Regulatory Acceptable Concentration (Toxicity value divided by a safety factor respectively trigger value
RAC_{dyn}	RAC used in the dynamic approach considering exposure shorter than in the standard tests to derive the RAC
RS	Risk segment
RMS	Risk management segment
SC	Steering Committee
SSD	Species Sensitivity Distribution
Std	Standard deviation
TER	Toxicity Exposure Ratio
TK	Topographic map (Topographische Karte),, 1:25 000

TOA	Time of Application
TOR	Time of Reproduction
ToT, ToTh	Time over Threshold = Total time of PEC above RAC, calculated by the dynamic exposure model
TWA	Time Weighted Average
UBA:	Umweltbundesamt (Federal Environment Agency)

1 Summary

Background and objectives

The intention of the UBA Project 3707 63 4001 was to further work out the scientific basis and the possibilities for the implementation of a geo-referenced probabilistic risk assessment of plant protection products in Germany focussing on drift and volatization / deposition entries from permanent crops like vine, orchards and hops into edge-of-field water bodies. The general aim of the new approach is to achieve a more realistic risk assessment compared to the current worst case scenario approach. As a consequence, the approach would enable the reduction of the current maximum application distance from 20 m to 10 m, nevertheless ensuring the protection of non-target species. Therefore, one central component of the approach is the identification of "hotspots" (areas with a high risk for surface water organisms due to pesticide entries) and to reduce these risks to acceptable levels by local risk mitigation measures.

In a previous project of the UBA (Schulz et al. 2007, 2009) the basis for the geodata based probalistic approach was layed down and open questions were identified. Therefore, the comprehensive project objectives were:

- To decide on the model assumptions and parameters used to calculate the surface water concentrations (PEC values) due to drift and volatilization/deposition entries
- To provide a technical concept for a realisation of this PEC estimation in GIS
- To collect ecological background information for trait based criteria allowing to identify ecologically critical aggregations of water body segments with PEC surpassing the current RAC (hotspots)
- To identify such hotspots for water bodies close to permanent crops in Germany
- To evaluate the new approach under ecological, socio-economic, and regulatory aspects
- To compare the approach with other regulatory approaches
- To propose a strategy for implementing the approach in practice
- To give an outlook on the possibilities of transferring the new concept to other routes of pesticide entry into water bodies (i.e. runoff and drainage) and field crops, and
- To provide a framework document for describing the new approach and a manual covering the technical aspects.

Project organisation

The consortium consisted of six partner institutions and worked together using telephone conferences, emails and google groups, and eight project meetings. An intermediate report was delivered in July 2009. The work was organised in six different work packages which were reflected by the report structure. An advisory board was formed with further experts and stakeholders, and the progress of the project was discussed intensively at two meetings. Furthermore, a three-day workshop with 34 participants from research, administration and industry was held at the UBA in Dessau in November 2009 (a workshop report is available as appendix C of this report).

Results

At a glance the following results were achieved in the project:

- The suggested **modelling approach and parameters to calculate the drift and volatilization / deposition inputs** were documented based on the open literature, project reports and discussions during the project meetings and the workshop. Entries are calculated for water body segments of 25 m length up to a distance of 150 m to crop areas depending on the distance between crop and water body and drift mitigation by vegetation and application technique. Variability is considered for the wind direction at the time of application and deposition rates (based on Rautmann drift table data which cover also the variability of wind speed during application). In order to achieve a total level of protection of the aquatic population of at least 95 % and considering that some assumptions of the approach are still conservative, it was decided to use the 90th centile of the calculated probability distributions of pesticide entries for each segment for the further calculations (Chapters 4.2 and 4.3).
- With **the static exposure model**, initial concentrations (PEC_{ini}) for a given entry only depend on the geometry of the water body. For small lentic water bodies and the shore line of large rivers, channels, lakes and ponds the 'standard ditch model' was used (lentic water body of 1 m width and 30 cm water depth (Chapter 4.4).
- To consider the fact that most edge of field water bodies are neither lentic nor characterized by 30 cm water depth a completely **new exposure model was developed for lotic waters**. The new approach uses, among others, the hydro-dynamic parameters water depth and stream velocity and therefore considers pesticide transport and dilution with time. The model calculates a topologically correct stream model, allowing for a near-to-reality PEC_{ini} calculation as a function of time and stream flowpath (Chapter 4.6). An important result of these calculations was that due to a minor water depth often found in small ditches in reality the resulting PEC_{ini} values were higher compared to the current PEC_{ini} values based on the standard ditch model. On the other hand, the local $PEC_{TWA(1h)}$ values calculated by the new dynamic approach were significantly lower because of the consideration of the flow velocity.
- For the implementation of site-specific **PEC_{ini} calculations within GIS**, a **geo-referenced data base** was established containing the relevant input parameters and the resulting centiles of the calculated pesticide entry distribution for each water body segment (Chapter 5). In addition, a **web-based program for the calculation of PEC_{ini}** on the basis of the static surface water model was developed. This tool enables the calculation and visualization of potential risk segments and, considering the hotspot criteria (see below), also calculating the segments along surface waters which have to be managed to reduce the PEC_{ini} (management segments).
- From an ecological point of view, single segments with a PEC exceeding the RAC may be of minor importance for the sustainability of a population. Thus, the spatial aggregation of such 'risk segments' was evaluated. A previously proposed generic **hotspot criterion** (UBA 2007) was refined based on a literature review of recovery and recolonisation studies, trait based identification of realistic worst case (macroinvertebrate) species, and population modelling (Chapter 6.2). **Tolerable effects for lethal effects** were calculated for these realistic worst case species under the prerequisite that after a 10 year application period the population abundance is within a range of +/- 20 % of the control abundance. Based on these analyses it was concluded that for a generic risk assessment the original UBA criterion should be used: **Along 1000 m stream (respectively ditch or shore line) the total effect on a population should not exceed 10 %**. For **multiple applications** suggestions of toler-

able effects of the single application events depending on the total tolerable effect and the number of applications per season were made (Chapter 6.4).

- In order to **calculate the expected effect from the PEC of a water body segment**, it is suggested to consider the RAC as the EC_{10} of a **logistic dose-response function with a realistic worst case slope** of 4. In addition, the often very short exposure in lotic water should be taken into account. Therefore, a literature review of acute effects related to exposure duration was conducted and realistic worst case estimations to **adapt the RAC to pulse exposure conditions** were developed (Chapter 6.3).
- For a product specific assessment **higher tier exposure and effect assessments** offer the option to refine the generic hotspot criteria (i.e. the tolerable effect level and the relevant spatial scale) and effect estimations (e.g. slope of the dose-response curves, effects of pulse exposure).
- The **identification of potential hotspots** and thus of water body segments where local risk mitigation measures are necessary was **based on a generic worst case product** applied with 1 kg ha-1 and assumed to be only authorized under the current registration scheme with the maximum label restriction related to the distance between application and surface water. Thus, the RAC would be just equal the PEC calculated for 20 m distance. Based on this assumption, the surface water segments where the RAC will be surpassed upon reduction of the maximum application distance from 20 to 10 m were identified using the geo-referenced probabilistic exposure calculations and the generic hotspot criterion. An additional advantage of this generic calculation is that the estimated PEC values can be used also for product related assessments, because the PEC_{ini} and $PEC_{TWA(1h)}$ from drift or volatilization / deposition entries are not dependent on any product properties except the GAP. Thus, PEC resulting from a single application are directly proportional to the application rate.
- The generic hotspot calculations carried out with the **static surface water model resulted in 2057 km management segments** for all areas with vine, hops and orchards in Germany (except the region ‘Altes Land’, Chapter 7).
- For the **dynamic exposure model**, only calculations for two single example streams in the hops region Hallertau were possible within this project. For these examples it was assumed that all fields situated along a stream were treated with a pesticide during a period of 2 days under realistic worst case stream conditions (low flow velocity and small water depth). The duration of exposure was considered by the adaption of the RAC (Chapter 6.3). Under these assumptions it was shown that the predicted effects on aquatic organisms were significantly reduced compared to the predictions based on the static model. Since the two streams were shown to be representative regarding their hydrodynamic parameters for streams in this region and among the ones with the highest exposure potential, the results were extrapolated to the hop regions in Germany and to all permanent crop areas in Germany (except for the ‘Altes Land’). In this way, about **200 km of management segments can be expected** for lotic waters in all vine, hops and orchard areas in Germany (Chapter 8).
- The **use of the dynamic flow model is proposed for further use** because it allows a more realistic estimation of the exposure situation and thus the effects on aquatic organisms. On the other hand, the dynamic model needs more research and development to allow a **Germany-wide hotspot calculation**. When the necessary data for application of the dynamic model for all relevant water bodies are available, the technical implementation of

the new method including the refinded hotspot criteria into the risk assessment for pesticides is possible within an intermediate time frame.

- The most reasonable **way to organize a management of the generically identified hotspots** is to establish a control board chaired by the BVL with participants from the UBA, JKI and heads of the federal agricultural services. This group, supported by a scientific institute with profound GIS knowledge, should organise the mitigation process in close cooperation with the local actors and apart from the pesticide use authorization process. This should be affordable with reasonable cost in a five-year period (Chapter 9).
- Thereafter, as decision criterion for the authorization of plant protection products, the consortium recommends that **the use of the product should not result in new hotspots**.
- Thus, **nation wide hotspot identification and management is considered to be essential for the implementation of the approach** because otherwise the protection of the local population cannot be ensured. However, the hotspot management should **not be part of the authorization process** but embedded within national intiatives, such as the 'Nationaler Aktionsplan zur nachhaltigen Anwendung von Pflanzenschutzmitteln (NAP)'.
- In principal, the new model approaches and the results of this project are considered to be **transferable to other entry routes and also to arable crops** (see Chapter 10). However, one should have in mind that an application on the entire area of field crops in Germany would need much more data and calculation time compared to the hops, vine and orchard areas.

Conclusion

The most important advantage of the proposed GeoRisk approach is that on the basis of **a more realistic geo-referenced risk assessment** the efforts in risk management/ mitigation could be more focussed on those localities along affected water bodies near treated agricultural areas which are potentially at highest risk due to the expected exposure situation (**hotspot management**). Consequently, such a spatially differentiating risk management could also open up the **possibility of decreasing spraying restrictions** for farmers to a necessary minimum while maintaining a high protection level for the aquatic environment.

However, due to the complexity of the newly developed dynamic exposure model and missing data for several input parameters it was not possible within this project to reach a stage in the implementation of the dynamic model which would allow for a nationwide application of this modelling approach.

Prior to a nation wide implementation of the approach the following steps are recommended:

1. To start a joint elaboration of the parameters required for the determination of the hotspots based on the dynamic model for a more realistic simulation of lotic water bodies
2. To calculate the management segments for a pilot study area (e.g. the Hallertau) and, if possible, at least one further area with a different permanent crop, and
3. To perform a pilot project, such as a field test e.g. in the Hallertau as a clearly defined area characterised by one specific culture, including hotspot management, use of the new mitigation measures and a chemical and ecological monitoring.

2 Zusammenfassung

Hintergrund und Zielsetzung

Die Intention des UBA-Projektes 2707 63 4001 (GeoRisk) war, die wissenschaftliche Basis und die Möglichkeiten für die Einführung einer georeferenzierten probabilistischen Risikobeurteilung für die Anwendung von Pflanzenschutzmitteln in Deutschland zu erarbeiten. Dabei lag der Fokus auf Drifteinträgen aus Dauerkulturen (Hopfen, Wein, Obst) in Oberflächengewässer in der Nähe der Kulturen.

Übergeordnetes Ziel dieses neuen Ansatzes ist es, eine realitätsnähere Risikoanalyse als mit dem zurzeit verwendeten Worst-Case-Szenario-Ansatz zu ermöglichen. In der Konsequenz würde dies die Vereinfachung und Verringerung der heutigen Abstandsauflagen ermöglichen, ohne den Schutz der aquatischen Populationen zu verringern. Um dies zu erreichen, ist die Identifikation von „Hotspots" (Gewässerstrecken mit einem ökologisch kritischen Eintragsrisiko) und deren Management durch lokale Risikominderungsmaßnahmen eine zentrale Komponente des GeoRisk-Ansatzes.

In einem vorherigen UBA-Projekt (Schulz et al. 2007) wurde die Grundlage für einen georeferenzierten probabilistischen Ansatz gelegt und noch offene Fragen identifiziert. Die einzelnen Ziele dieses Projektes waren demnach:

- festzulegen, welche Modellannahmen und welche Parameter verwendet werden sollen, um die Konzentration von Wirkstoffen in Oberflächengewässern als Folge von Einträgen durch Drift und Verflüchtigung / Deposition zu berechnen,
- ökologische Hintergrundinformation zu sammeln, um Trait-basierte Kriterien zu entwickeln, die eine Identifizierung ökologisch kritischer Häufungen von Gewässerabschnitten mit hohen PEC-Werten erlauben,
- die potentiellen Hotspots für Gewässer in der Nähe von Dauerkulturen zu identifizieren,
- die technische Implementierung eines Tools zu realisieren, mit dessen Hilfe eine georeferenzierte und probabilistische Risikobeurteilung möglich ist,
- den vorgeschlagenen Ansatz unter ökologischen, regulatorischen und sozioökonomischen Gesichtspunkten zu bewerten,
- die vorgeschlagene Vorgehensweise mit dem zur Zeit verwendeten Beurteilungsverfahren sowie anderen in der EU verwendeten Methoden zu vergleichen,
- eine Strategie vorzuschlagen, den Ansatz für die Risikobewertung von Drift- und Verflüchtigungs-/Depositionseinträgen von Dauerkulturen in Deutschland umzusetzen,
- die Möglichkeiten der Übertragbarkeit des Ansatzes auf andere Eintragspfade und auf Feldkulturen zu diskutieren, und
- ein Rahmenkonzeptpapier mit der Beschreibung des vorgeschlagenen Ansatzes sowie einen technischen Leitfaden zum entwickelten webbasierten Tool zur Verfügung zu stellen.

Projektorganisation

Das Projektkonsortium bestand aus insgesamt 16 Wissenschaftlern aus sechs Instituten, die mit Hilfe von Telefonkonferenzen, Emails, Google-Groups und acht Projekttreffen zusammenarbeiteten. Die Aufgaben waren in sechs verschiedenen Arbeitspaketen organisiert, was sich auch im Aufbau des hier vorliegenden Berichtes widerspiegelt.

Für das Projekt wurde ein wissenschaftlicher Beirat mit weiteren Experten gebildet, mit dem der jeweilige Projektfortschritt auf zwei Beiratssitzungen diskutiert wurde. Weiterhin wurde im November 2009 ein dreitägiger Workshop im Umweltbundesamt in Dessau mit insgesamt 34 Teilnehmern aus Forschung, Behörde und Industrie organisiert (der Workshop-Bericht ist als Anhang C dieses Berichts verfügbar).

Ergebnisse

Die folgenden Ergebnisse wurden im Projekt erzielt:

- Der vorgeschlagene **Modellierungsansatz und die Modelparameter** zur Berechnung der Einträge durch Drif bzw. Verflüchtigung und Deposition wurden auf der Basis von Veröffentlichungen, Projektberichten und Diskussionen innerhalb der Projekttreffen und des GeoRisk-Workhops festgelegt und dokumentiert (Kapitel 4.2 und 4.3).
  PEC-Berechnungen erfolgen für alle Gewässersegmente von 25 m Länge im Bereich von 150 m um Kulturflächen als Funktion des Abstandes zwischen Gewässer und Kultur sowie Eintragsminderungsfaktoren wie Applikationstechnik oder abschirmende Vegetation. Die Variabilität von Einträgen wird in Bezug auf die Windrichtung zum Zeitpunkt der Applikation und die Depositionsrate (auf der Basis der BBA/JKI-Driftversuche), welche auch die Variabilität der Windgeschwindigkeit zumindest teilweise abdecken) mit Monte-Carlo-Simulationen berücksichtigt. Für die weiteren Berechnungen wurde das 90. Zentil der abgeleiteten Häufigkeitsverteilung der Einträge in jedes Segment (lokale Eintragsverteilung) verwendet. Wegen weiterer im Ansatz enthaltenen konservativen Annahmen wurde dies als ausreichend angesehen, insgesamt das angestrebte Schutzniveau von 95 % (Schutz der lokalen Populationen mit 95 % Sicherheit) zu erreichen.
- Im **statischen Expositionsmodell** (Kapitel 4.4) sind die initialen PEC-Werte (PEC_{ini}) für einen gegebenen Eintrag nur abhängig von der Geometrie des Gewässers. Für kleinere Gewässer (bis 3 m Breite) und die Uferlinie von größeren Flüssen sowie von Seen und Teichen wurde das „Standardgrabenmodell“ verwendet (Stehgewässer von 1 m Breite und 30 cm Tiefe).
- Um zu berücksichtigen, dass die meisten Gewässer in der Nähe landwirtschaftlicher Flächen weder stehend sind noch eine Wassertiefe von 30 cm aufweisen, wurde ein vollständig neues **Expositionsmodell für Fließgewässer** entwickelt (Kapitel 4.6). Dieses Modell verwendet unter anderem die hydrodyamischen Parameter Wassertiefe und Fließgeschwindigkeit und damit Transport und Verdünnung der Wirkstoffe im Gewässer, und setzt daher die Berechnung eines topologisch korrekten Gewässermodells voraus. Neben den hydrodynamischen Parameter wird auch die Stochastizität der Eintragsereignisse im Oberlauf eines Segments berücksichtigt. Der Ansatz erlaubt somit eine realitätsnahe Berechnung der PEC-Werte als Funktion von Zeit und Fließstrecke. Das wesentliche Ergebnis von Berechnungen mit dem dynamischen Modell ist, dass einerseits die berechneten PEC_{ini}-Werte wegen der oft geringen Wassertiefe der Gewässer höher sind als im bisher verwendeten Standardgewässermodell. Andererseits sind durch die Strömung die gemittelten Konzentrationen über 1 Stunde ($PEC_{TWA(1h)}$) der einzelnen Gewässerabschnitte deutlich geringer.

- Für die Implementierung der lokalen PEC-Berechnungen wurde eine **georeferenzierte Datenbank** der relevanten Eingangsparameter und der sich daraus ergebenden Zentile für die Drifteinträge in die einzelnen Gewässerabschnitte erstellt (Kapitel 5). Weiterhin wurde ein **Web-basiertes Programm zur deutschlandweiten Berechung der initialen PEC-Werte auf der Basis des statischen Gewässermodells** entwickelt, implementiert und dokumentiert (Kapitel 5 und Anhang B). Dieses Tool erlaubt die Berechnung und Visualisierung von potentiellen Risikosegmenten und die Identifizierung von Hotspots und Managementsegmenten.
- Aus ökologischer Sicht kann die Überschreitung der Regulatorisch Akzeptablen Konzentration (RAC) in einzelnen isolierten Segmenten für den Schutz der Populationen von geringer Bedeutung sein. Daher wurde die Bedeutung von räumlichen Häufungen solcher Risikosegmente für die zu schützenden Populationen analysiert. Ein vorher vom UBA entwickeltes **Hotspot-Kriterium** (UBA 2007) wurde auf der Basis veröffentlichter Fallstudien zur Wiedererholung von Population nach Stress, Traitdatenbanken zu relevanten Eigenschaften (z. B. Verbreitungspotential, Entwicklungsdauern, Wachstumsraten u. a.) von Arten, Monitoringstudien und Populationsmodellierung überprüft und weiter entwickelt (Kapitel 6.2). Für Taxa mit hohen Populationswachstumsraten wie z. B. Phytoplankton und Zooplankton ergab eine Literaturstudie zum Wiedererholungspotential, dass mit hoher Sicherheit auch nach 90 % Abundanzreduktion eine Wiedererholung innerhalb eines Jahres zu erwarten ist. Anhand ihrer Traits wurden „Realistic Worst-Case-Arten“ von Makroinvertebraten bestimmt: die Muschel *Anodonta cygnea*, die Libelle *Calopteryx virgo* und der Oligichaet *Dero digitata.* **Tolerierbare lethale Effektstärken** wurden für diese Arten mit Hilfe von Populationsmodellen unter der Annahme abgeleitet, dass nach 10jähriger Anwendung eines Produktes die Abundanz noch im Beriech von +/- 20 % der Kontrollpopulation sein soll. Im Ergebnis sollte für eine generische Risikoabschätzung folgendes Kriterium verwendet werden:
  Auf 1000 m Gewässerstrecke darf der Effekt einer Applikation auf die gesamte Population nicht stärker als 10 % sein.
  Für Fische und Makrophyten wird die Beibehaltung des 10 %-Kriteriums empfohlen, da für Wirbeltiere höhere Schutzanforderungen angenommen werden und Makrophyten als Schlüsselarten in aquatischen Systemen angesehen werden.
- Um die aus der PEC **zu erwartende Effektstärke** in einem Gewässersegment zu berechnen, wird die **Verwendung einer logistischen Dosis-Wirkungskurve** mit der RAC als EC_{10} vorgeschlagen. Eine realistische Worst-Case-Steigung der Kurve (slope = 4) wurde aus Daten für Carbaryl abgeleitet. Zur realistischeren Berücksichtigung der in Fließgewässern oft nur sehr kurzen Expositionen (< 1 h bis wenige h) wurden empirische Formeln zur Anpassung der aus Standardtests abgeleiteten RAC an Pulsexpositionen entwickelt (Kapitel 6.3).
- Für **multiple Applikationen** wurden Vorschläge für **angepasste Effektschwellen** für die einzelnen Applikationsereignisse in Abhängigkeit von der Anzahl der Applikationen und dem insgesamt tolerierbaren Effekt gemacht (Kapitel 6.4).
- Für verfeinerte Risikoanalysen im Rahmen der Zulassung spezifischer Produkte können, wie bisher, **Higher-Tier-Ansätze** auf der Expositions- (z.B. Berücksichtigung der Dissipation) und der Effektseite (z.B. Verfeinerung der Hotspot-Kriterien für die relevanten Arten, Berücksichtigung spezifischer Informationen zu Dosis-Wirkungsbeziehungen und Effekten von Pulsexposition) angewandt werden.

- Die Identifizierung von potentiellen Hotspots und somit Gewässersegmenten, an denen lokale Risikominderungsmaßnahmen durchgeführt werden sollten, wurde für ein generisches Worst-Case-Produkt mit einer Aufwandrate von 1000 g/ha durchgeführt. Dabei wurde angenommen, dass dieses Mittel nach dem jetzigen Verfahren in Deutschland nur mit der größten Abstandsauflage gerade noch zugelassen werden kann. In diesem Fall ist die RAC gleich der PEC, die sich für 20 m Abstand ergibt. Unter dieser Annahme wurden mittels der georeferenzierten probabilistischen Expositionsberechnung die Gewässersegmente bestimmt, in denen bei einer Abstandsreduzierung von 20 m auf 10 m die RAC überschritten würde (Risikosegmente).
  Diese generische Analyse bietet - neben der Hotspot-Identifizierung – den Vorteil, dass die lokalen Eintrags- bzw. PEC-Verteilungen auch für produktspezifische Berechnungen verwendet werden können, da sie für einzelne Applikationen nur von der Aufwandmenge (aber nicht von Substanzeigenschaften) abhängen.
  Durch Anwendung des generischen Hotspot-Kriteriums können ökologisch unkritische einzelne Risikosegmente aus der weiteren Betrachtung ausgeschlossen und so die Managementsegmente bestimmt werden.
- Mit dem **statischen Expositionsmodell**, angewandt auf alle relevanten Gewässer, wurden für alle Hopfen-, Obst- und Weinanbaugebiete in **Deutschland** (ausgenommen: Altes Land) **insgesamt 2057 km Managementsegmente** berechnet (Kapitel 7).
- Mit dem neu entwickelten **dynamischen Expositionsmodell** konnten im Rahmen des Projektes nur für **zwei Beispielbäche in der Hopfenregion Hallertau** Berechnungen durchgeführt werden (Kapitel 8). Für diese Berechnungen wurde angenommen, dass alle Flächen entlang des Baches in einem Zeitfenster von 2 Tagen behandelt wurden. In Bezug auf die Hydrodynamik wurden realistische Worst-Case-Bedingungen angenommen (niedrige Fließrate, niedriger Wasserstand). Die Expositionsdauer wurde bei der Effektbewertung durch Anpassung der RAC berücksichtigt (s. Kapitel 6.3). Unter diesen Annahmen konnte gezeigt werden, dass die vorhergesagten Effekte auf die Populationen signifikant geringer waren, also die auf der Basis des statischen Expositionsmodells.
  Die für repräsentative Gewässer erzielten Ergebnisse konnten zu einer Hochrechnung der zu erwartenden Managementsegmente für die Hopfenanbaugebiete herangezogen werden. Diese Analyse berechnete 11,6 km Managementsegmente. Verglichen mit den Berechnungen der „State of the Art-Methode“ waren das 8 %.
  Übertragen auf alle Sonderkultur-Anbaugebiete in Deutschland (ausgenommn: Altes Land) führt das zu der vorsichtigen Einschätzung, dass **unter den hier gewählten Voraussetzungen für die Berechnung deutschlandweit mit ca. 200 km potentiellen Managementsegmenten für die Sonderkulturen** gerechnet werden kann.
- Für die Zukunft wird die Anwendung des dynamischen Expositionsmodells für Fließgewässer empfohlen, da er eine realitätsnähere Abschätzung der Exposition und somit der Effekte ermöglicht. Auf der anderen Seite benötigt dieser Ansatz noch weitere Arbeiten, bevor er bundesweit angewendet werden kann. Wenn die benötigten hydrodynamischen Daten für alle relevanten Gewässer abgeleitet sind, ist eine technische Implementierung des dynamischen Ansatzes inklusive der verfeinerten Hotspot-Kirterienfür die bundesweite Anwendung jedoch mittelfristig möglich.
- Für die **Umsetzung des Managements** der generisch indentifizierten Hotspots sollte ein **Kontrollgremium** unter Vorsitz des BVL ins Leben gerufen werden, in dem UBA, JKI und

die Landwirtschaftlichen Dienste der Länder vertreten sind. Dieses Gremium sollte, mit Unterstützung mit profunder GIS-Kenntnis, die Risikominderungsmaßnahmen in enger Zusammenarbeit mit den lokalen Akteuren organisieren. Dies sollte mit zumutbaren Kosten innerhalb von 5 Jahren nach der Identifizierung der generischen Hotspots möglich sein (Kapitel 9.3).

- Bei Einführung des GeoRisk-Ansatzes sollte die **Zulassung von Pflanzenschutzmitteln** darauf beruhen, dass **durch die beantragte Nutzung des Mittels keine neuen Hotspots zu erwarten** sind.
- Eine erfolgreiches **bundesweites Hotspot-Management ist somit die Voraussetzung für die Einführung einer neuen Risikobewertung** von Pflanzenschutzmitteln mit vereinfachten und reduzierten Auflagen. Das lokale Hotspot-Management sollte jedoch **nicht Teil des Zulassungsverfahrens** selbst sein, sondern in nationale Initiativen wie beispielsweise den Nationalen Aktionsplan zur nachhaltigen Anwendung von Pflanzenschutzmitteln (NAP) eingebettet werden.
- **Prinzipiell ist der vorgestellte Ansatz auch für andere Eintragspfade wie Run-off und Drainage und damit auch auf Feldkulturen anwendbar**. Im Detail werden aber zurzeit Probleme gesehen, da z. B. die Berechnung von Einträgen über Run-off komplexer ist als für Drift und für Flächenkulturen eine viel größere Datenmenge als für Raumkulturen zu bewältigen wäre (Kapitel 10).

Schlussfolgerungen

Der wichtigste Vorteil des vorgeschlagenen GeoRisk-Ansatzes liegt darin, dass durch die **realitätsnähere georeferenzierte Risikoanalyse** das Risikomanagement auf solche Gewässerabschnitte fokussiert werden kann, die für die Vermeidung nicht akzeptabler Effekte auf aquatische Populationen am wichtigsten sind, nämlich die mit potentiell hohen Einträgen von Pflanzenschutzmitteln (**Hotspot-Management**). Als Folge eines solchen räumlich differenzierten aber produktunabhängigen Risikomanagements ergibt sich die Möglichkeit, produktbezogene Anwendungsauflagen auf ein notwendiges Minimum zu reduzieren, welches den Schutz der Gewässer weiterhin gewährleistet.

Wegen der Komplexität des neu entwickelten dynamischen Ansatzes der Expositionsmodellierung und der noch fehlenden Datenbasis für einige notwendige Eingangsparameter konnte innerhalb des Projekts noch keine bundesweite Anwendung dieses Ansatzes realisiert werden.

Um den hier vorgestellten Ansatz einer georeferenzierten probabilistischen Risikoanalyse bundesweit für Sonderkulturen in Deutschlang einzuführen, werden folgende Schritte vorgeschlagen:

1. Ausarbeitung der Datenbasis für eine generische Hotspot-Analyse mit dem dynamischen Expositionsmodell für Fließgewässer in alle Sonderkulturregionen in Deutschland
2. Identifizierung der Managementsegmente für ein Pilotgebiet (z. B. die Hallertau) und möglichst zumindest ein weiteres Gebiet mit anderer Kultur
3. Durchführung eines Pilotprojektes, z. B. in der Hallertau, als klar abgegrenztes durch eine Kultur charakterisiertes Gebiet, inklusive Hotspot-Management, reduzierter Anwendungsauflagen sowie chemischem und biologischem Monitoring.

3 Introduction

Udo Hommen, Roland Kubiak

3.1 Background

Currently the aquatic risk assessment for the authorisation of plant protection products in Germany – similar to the approach in the EU – is based on a realistic worst case scenario approach: The expected concentration of the active substance in edge of field water bodies (PEC = Predicted Environmental Concentration) considers entries via drift, volatilization and deposition, run-off and drainage by means of different exposure models based on conservative assumptions with respect to the environmental conditions. In general it is assumed that during application the wind is always blowing from the treated field in direction of the nearest water body and that there is no drift mitigating vegetation between the crop and the water body. The water body is assumed to be a static ditch of 1 m width and 1 m length.

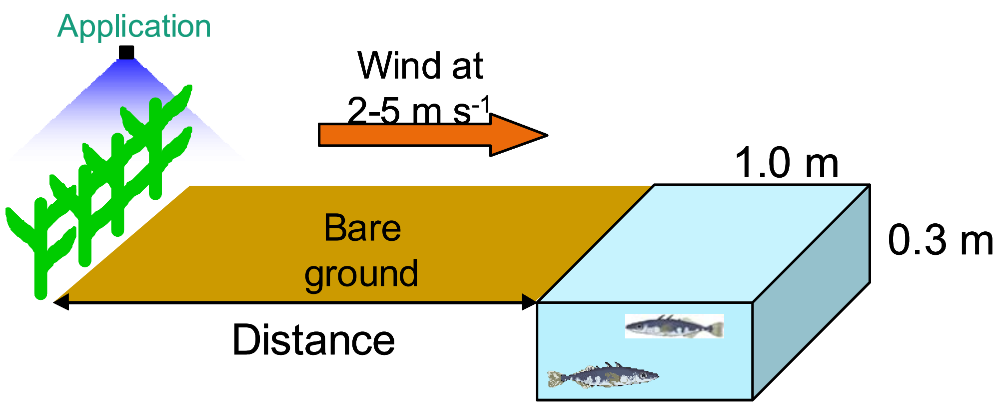


Figure 3-1: The current scenario for estimating PECs due to drift entries

The resulting PECs are then compared to the effect concentrations derived from ecotoxicological studies using the TER approach. For the acute risk assessment for fish, as an example, the TER would be:

$$TER = 96\text{ h} - LC_{50fish} / PEC_{ini}$$

The TER is compared to specific trigger values (i.e. 100 in the case of acute risk assessment), and the risk resulting from the application of the plant protection product is considered acceptable if the TER is above the given trigger.

To achieve acceptable risk, different risk management options are available, e.g minimum distances for the application close to water bodies or the use of drift reducing spraying equipment. Thus, based on the recommended use of the product, its physico-chemical properties and its ecotoxicity, different mitigation measures have to be applied to avoid unacceptable effects by using this specific product. However, these label instructions can become very complex and thus, they are often considered not to be practicable by the farmers. Furthermore, they are difficult to control by the local authorities.

Due to the different worst case assumptions involved in the derivation of the mitigation measures (e.g. wind direction, no mitigation by vegetation, static water bodies), they are often considered to be overprotective and consequently not very well accepted by the farmers; furthermore, they are difficult to control by the local authorities.

On the other hand, the current approach includes also elements which are not conservative; e.g. the assumption that all water bodies are 30 cm deep is obviously not protective for more shallow streams and ditches. Due to the fact that in the recent risk assessment approach assumptiosn with different degrees of conservatism are combined, the finally achieved level of protection therefore is not well defined.

As a consequence, over the past years there was an increasing interest by farmers, the plant-protection industry and regulators to develop a more realistic risk assessment which should allow simplified and reduced product specific risk mitigation measures while maintaining the existing level of protection for the environment.

3.2 Development of a geo-data based probabilistic approach in Germany

As an alternative to the current approach based on worst case point estimations the use of geo-data and statistical distributions has been discussed for several years in Germany to better consider the spatial and temporal variability as well the uncertainty of parameters driving exposure and effects of pesticides (e.g. Golla et al. 2002, Klein et al. 2006, UBA/BVL/BBA/IVA 2006, Schulz et al. 2007, 2009, Trapp & Thomas 2008).

Geographical Information Systems (GIS) serve as the basis to consider the spatial variability, i.e. the location of water bodies in relation to agricultural areas: Instead of a virtual worst case scenario the local scenarios in the real landscape can be considered, i.e. the calculation of the PEC for water body segments of 25 m lengths.

For each of the water body segments, the variability (natural stochasticity) and uncertainty (ignorance or measurement errors) of the input parameters for the exposure model can be considered by Monte-Carlo simulations and result in a local distribution of entries or PECs (see figure 3-2). By taking a specific percentile from each of these local distributions a new distribution describing the spatial variability can be derived. This type of distribution is called a landscape level distribution in this report (PEC distribution at a national level in figure 3-2, but it could of course also derived only for a specific region).

Different approaches have been suggested in the past describing how this new spatial information could be used.

One suggestion (e.g. Klein et al. 2006) was to determine a spatial distribution of the PECs from all relevant water bodies segments for the specific crops and then use for example the 90^{th} or 95^{th} percentiles of these landscape level distributions to determine the necessary risk mitigation measures. By doing this, a step back from the potential spatially explicit resolution of data to a full probabilistic approach is made. Thus, any spatial information on the combination of parameters driving the local exposure situation is lost and - for example - the 10 % or 5 % of water bodies with PEC-values exceeding the RAC (Regulatory Acceptable Concentration) are not further considered. This is hardly in agreement with the protection aim of protecting local non-target populations.

Based on a proposal by the BBA (today JKI, BBA 2006) to implement a GIS-based probabilistic approach for the risk assessment of drift entries from permanent crops in water bodies and other studies (reported later inTrapp & Thomas 2008, 2009, Trapp et al. 2008), UBA, BVL, BBA and IVA have developed a framework document describing how the approach could be implemented in the execution of the German Plant Protection Act (UBA/BVL/BBA/IVA 2006).

In a research and development project of the UBA (UBA FKZ 206 63 402) this concept has been evaluated and refined – mainly based on an expert workshop at the UBA in 2007– by the University of Koblenz-Landau (Schulz et al. (2007, 2009). Now the spatial information on water body segments with high exposure potential is explicitely considered by identification of critical spatial aggregations of such risk (active management areas, AMA, in Figure 3-2, 'hotspots' within this report). In the hotspots, a local landscape-related risk management could be conducted to reduce the exposure in the segments with high PEC-values. This would reduce the risk for all products used there and product-specific mitigation measures could be simplified and reduced.

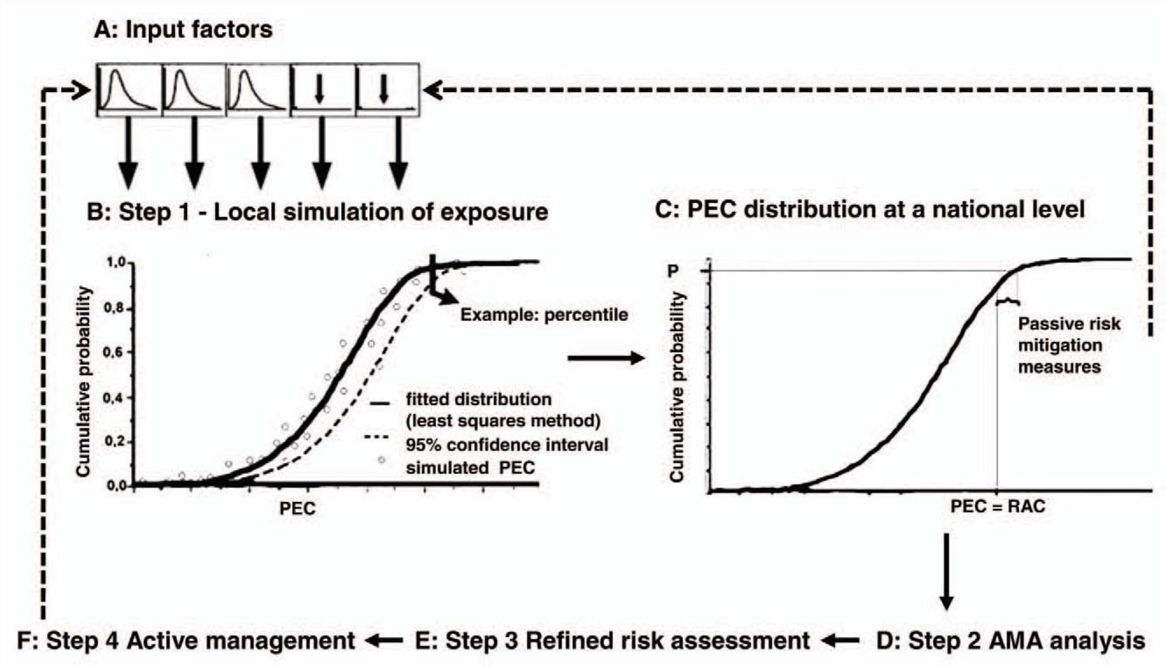
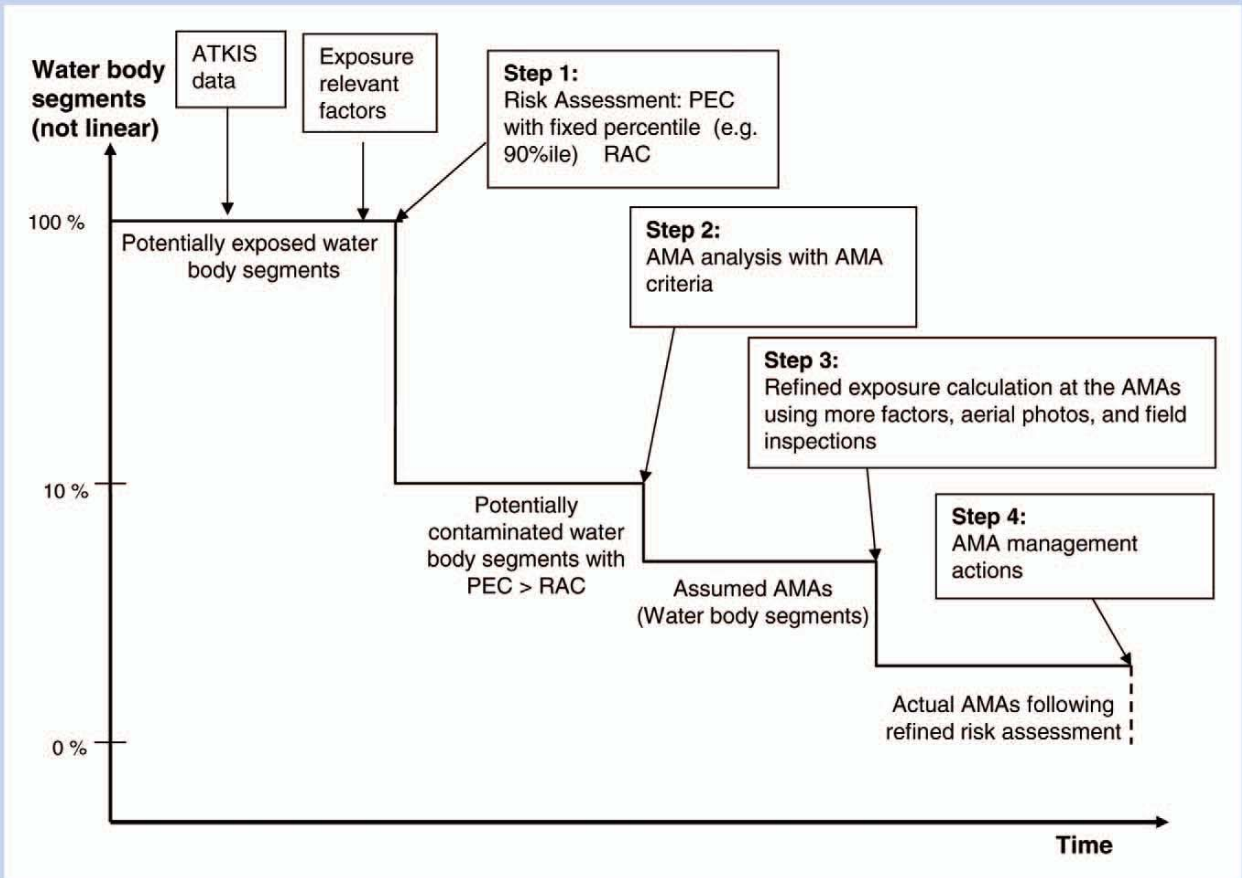


Figure 3-2: Scheme illustrating how and at which steps the setting of percentiles of exposure distributions occurs. The diagram also indicates that the number of active management area (AMA) man-agement measures determined through a feedback step the setting of percentiles and confi-dence limits (from Schulz et al. 2009)

In the final report of the project (Schulz et al. 2007) suggest a four-step approach (see also Figure 3-3):

1. Risk assessment for all relevant water bodies in Germany by means of geo-data and probabilistic methods.
2. Hotspot analysis considering the spatial aggregation and extension as well as the magnitude of exposure and the tolerable effects for populations.
3. Refined exposure estimation by means of aerial photographs or other high resolution data.
4. Risk management with a focus on landscape-related drift mitigation measures which results in a long-term reduction of risks for the aquatic populations.



5.
Figure 3-3: 'Principle scheme of the geodata-based probabilistic risk assessment for pesticides at the current state of discussion. First exposure is calculated on the basis of ATKIS using predicted environmental concentration (PEC) distributions for each water body segment and at the landscape level. All water body segments with PEC> the regulatory acceptable concentration (RAC) are incorporated in active management area (AMA) analysis, and for the identified AMAs a refined exposure calculation is conducted. At the remaining AMAs, there is a need for management measures.' Figure and legend copied from Schulz et al. 2009).

Some open questions were identified at the workshop in 2007 and listed in the project report by Schulz et al. (2007, 2009); they should be answered before such an approach can be implemented in Germany.

3.3 Aims, objectives and products of the GeoRisk project

The overall aim of a geo-data based probabilistic risk assessment of plant protection products in Germany is to establish a more realistic assessment allowing simplified and reduced substance specific risk mitigation measures while maintaining the existing level of protection.

The objectives of the GeoRisk projects were:

- Providence of the scientific basis for introducing a geo-data based probilistic approach and clarification of the open points identified in the former projects.
- Evaluation of the ecological, regulatory and socio-economic consequences of the new approach.

The GeoRisk project focuses on drift entries from pesticide application in permanent crops (orchards, grapes, hops). Other entry routes and the potential use of the approach for field crops are only shortly discussed in chapter 10 of this report.

According to the project specification the main products of the GeoRisk projects are:

– a documentation of the model assumptions and parameter values including uncertainty and sensitivity analysis for the GIS-based exposure and risk assessment. This is included as chapter 4 of this report,

– a GIS-based tool including data base and data management tool for the exposure and risk estimations according to the elaborated assessment approach,

– a technical manual for the implementation of the suggested approach in GIS (appendix 2 of this report), and.

– a framework document including a description of the suggested geo-data based probabilistic approach for the risk assessment of pesticide drift entries in water bodies including the a concept for its implementation.

3.4 Structure of this report

Based on the list of tasks in the project specification the GeoRisk project was organized in six work packages:

1. Exposure model and input parameters
2. Technical realisation in GIS
3. Derivation of hotspot criteria
4. Evalulation and implementation options of the approach
5. Extrapolation to other entry routes and field crops
6. Workshop organisation and project management

The structure of this report follows the organisation in work packages: chapters 4 – 6 describe the models and parameters of the exposure model (WP 1, chapter 4), the development of the data base and model realisation in GIS (WP 2, chapter 5), the evaluation of the hotspot criteria (WP 3, chapter 6).

In chapter 7, the results of the static exposure model assuming that all water bodies can be considered as lentic with a depth/width ratio of 0.3 are summarized. Using the geodata based probabilistic drift entry calculation (chapter 4) and the generic hotspot criterion (chapter 6), the resulting number and lengths of risk and management segments are presented by differentiating the relevant water bodies for hops, vine and orchards in Germany.

In chapter 8 the dynamic exposure model described in chapter 4, which explicitly considers transport and dispersion in flowing waters and realistic hydrodynamic parameters including water depths, is applied to two representative streams located in the hops region Hallertau. Based on these examples an extrapolation to all relevant water bodies close to hop cultures in Germany is made to demonstrate the effects of the more realistic estimation on the number and length of hotspots to be managed.

In chapter 9 (corresponding to WP 4) the suggested approach is evaluated considering ecological, regulatory and socio-economic criteria. In addition, suggestions for the implementation of the GeoRisk approach for permanent crops in Germany and options for local risk mitigation measures are listed and discussed. Finally, the GeoRisk approach is compared to deterministic and scenario based approaches.

Options for a further refinement of the approach (e.g. including other entry routes) and extrapolation to field crops are discussed in chapter 10 (WP 5).

Chapter 11 gives more details on the project partners and their responsibilities, the communication in the project via telephone conferences and meetings and the advisory board meetings. The chapter also includes the summary of the GeoRisk workshop held in November 2009 at the UBA Dessau. The full workshop report has been sent to all workshop participants before and is given as an appendix to this report.

Chapter 12 contains a glossary of terms.

Some parts of the project report should be used as stand-alone documents and thus they are given as appendices:

A. Framework document on the implementation of the geo-data based probabilistic approach of pesticide risk assessment in Germany (subarea: drift and deposition entries into water bodies)
B. Technical guideline (manual) for the GIS tool
C. GeoRisk Workshop Report
D. Project related publications or manuscripts for publication

3.5 Acknowledgements

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We are also grateful to Michaela Nicklas from the Intitute for Agrarecology, RLP Agroscience, for checkin the draft report for typing errors, layout, missing references etc. and Brigitte Peine from the Fraunhofer IME for additional checking, editing and formatting the final report.

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4 Model assumptions and input parameters for the geo-data based probabilistic exposure estimation

Michael Klein, Burkhard Golla, Martin Bach, Matthias Trapp

4.1 Introduction

In the German assessment scheme for the registration of plant protection products the input route "drift" has already been considered for many years. The method of estimating the PECini, which is generally accepted among experts, is based on realistic worst case assumptions essentially dependent on the crop and distance specific drift percentiles combined with a constant water depth.

When transferring this rather simple approach into a geo-referenced probabilistic model an important aspect must be the evaluation of all processes and input parameters regarding the level of protection achieved with the current system and intended with the new system.

This analysis takes also into consideration the results of a pilot study sponsored by the Umweltbundesamt and performed by the Universität Koblenz-Landau (Schulz et al. 2007).

In this work package the necessary algorithms to transfer the current methods into routines that can be used in the new geo-referenced approach and to find representative values for the relevant model parameters are defined.

The protection goal is to guarantee that unacceptable effects on the population of the respective ecosystem can be excluded with a high level of certainty (e.g. 95 %). It is assumed that by use of the 90th centile of the local entry distribution this goal can be achieved due to conservative assumptions in other parts of the assessment (see section 4.8 for an overview on the assumptions used for the exposure estimation).

However, this concentration is influenced by many processes and parameters. Presently, many of them cannot be described adequately with regard to their distribution, which makes an exact calculation of this target concentration impossible. Therefore, the different parameters contributing to the calculation of surface water concentrations are either considered probabilistically, with geo-referenced information, or by point estimations (usually conservative assumptions according to current knowledge and availability). An overview about the different parameters and processes which are considered in this chapter is given in the following table.

Table 4-1: Parameters and processes influencing surface water concentrations via drift

Parameter/Process	Description	P^{+}	G^{++}
Wind speed	indirectly via deposition rate	-	-
Deposition rate	experimentally based distance dependent distribution	yes°	no
Wind direction	uniform distribution	yes	no
Water side vegetation	seasonal dependent conservative values	no	yes
Emerse vegetation	seasonal dependent conservative values*	no	yes
Water depth	calculated using water width (reflecting medium situation)	no	no
Nozzle technique	conservative values dependent on spraying system	no	no
Volatilisation deposits	conservative model (EVA) dependent on vapour pressure	no	no
Degradation	not considered (conservative approach)	no	no
Volatilisation losses	not considered (conservative approach)	no	no

($^{+}$P = probabilistic, $^{++}$G = geo-referenced,*only if respective information is available, ° 90th percentile proposed)

4.2 Calculation of spray drift entries into surface waters

In the context of Directive 91/414/EEC aquatic exposure assessment drift inputs are calculated based on the JKI (former BBA) spray drift data (EC, 2002; FOCUS, 2002). The data stem from field studies in Germany (Ganzelmeier et al. 1995, Rautmann et al. 2001) and have also been used as the basis for the 90th percentile drift rates used in the national risk assessment.

Each data point is a measurement of drift deposition on a horizontal surface, expressed as a percentage of the nominal application rate. Data sets are available for the crop groups arable, vines, orchards and hops representing different types of application. The dataset comprises different numbers of trials and measurements. The trials were conducted at various sites at different days over a period of several years (1989 to 1992 and 1996 to 1999) according to the BBA Guideline for measuring direct drift during pesticide application in the field (BBA, 1992). The guideline requires that data is collected at wind speeds below 5 m/s and that the wind is always vectored rectangular to the driving direction (+/- 30°). The application conditions, such as air temperature, wind speed, wind direction, nozzle, vehicle speed, relative humidity are recorded for each trial. The trial data (Ganzelmeier et al. 1995, Rautmann et al. 2001) are available upon request at JKI. A comprehensive documentation of the experimental layout are given in Ganzelmeier et al. (1995).

4.2.1 Parameter wind speed

As already mentioned, the wind speed is currently not considered as an explicit parameter when calculating drift entries. However, it is indirectly considered because the drift tables are based on experimental studies performed at different wind situations according to the BBA Guideline for conducting spray drift experiments (BBA, 1992). Even though the original data did not allow further derivation of explicit dependencies between wind and amount of pesticide deposited the protocol fixed at least the maximum wind speed (5 m/s) at which these experiments were performed.

If the wind speed was considered in the new geo-referenced assessment model it would only influence the initial load. As long as volatilisation out of the surface water is not a dominant process the wind speed will not influence the time course of concentrations in surface water.

4.2.1.1 Calculation of spray drift considering wind speed explicitly

Due to local turbulences it is generally difficult to accurately calculate drift entries considering wind speed explicitly, especially for lower wind speeds. Also the change of drop diameters during transport through the air makes exact calculations an extremely difficult procedure.

However, based on a box model with a simple, approximate description of these processes, the fundamental influence of wind speed on spray drift can be analysed. The box model can be considered to predict mean drift values, whereas more complex models would additionally estimate the significant distribution of the entry.

The key processes in the box model are the transport of droplets in wind direction and the deposition due to gravity. Whereas the transport of droplets in wind direction can be considered as independent on their size, the deposition velocity is strongly influenced by the drop diameter because of the atmospheric friction, as shown in Table 4-2.

Table 4-2: Deposition versus drop diameter [Davies 1966]

Drop diameter [µm]	Deposition [cm/s]
1	0.0035
10	0.3
50	7.2
100	25
200	70
500	200
1000	385

Basically, smaller droplets are produced in orchards than in field crops, resulting in smaller deposition rates and leading to higher drift percentiles in orchards compared to field crops.

Assuming a constant fraction of droplets reaching the surface independent of the actual concentration in the air an exponential decay of deposition by drift can be expected with increasing distance from the target area. This is expressed by the following equation which makes the fraction reaching the surface per distance unit a function of wind speed, drop diameter and the height of the spraying system.

$$k = s / [(v+s] * H)$$
$$C(d) = C_0 * \exp(- k * d)$$
$$D(d) = C(d) * k * A$$
$$D_p(d) = App * D(d) / 100$$

k: decline per distance unit (1/m)
s: deposition velocity of droplets (m/s)
A: area below the concentration (1 m²)
H: height of the spraying system (m)
u: wind speed (m/s)
C(d): concentration (mg/m³)
C_0: initial concentration (mg/m³)
d: distance to the spraying system (m)
D(d): deposition at distance d (mg/m²)
App application rate (mg/m²)
$D_p(d)$: drift percentile at distance d

This model will never calculate exact spray drift amounts, but it gives an impression about the influence of wind speed on the expected deposition in the neighbouring area.

4.2.1.2 Availability of wind speed

Wind speed is generally not available at a local scale (e.g. in the field). The German Weather Service (DWD) uses a model for estimating wind speed at 2 m height from data of 10 m height. The model is applied in arable crops (Löbmeier, in personal communication, 2010). For permanent crops the wind situation within the growing systems is influenced by the layout (e.g. direction of the fruit rows) and is not comparable to field crops. Investigations of wind speed within orchards are planned by the working group of Van de Zande (Wennecker, in personal communication, 2009). Results are not available yet.

Instead, information could be made available from local weather stations. However, due to the strong time variations of this parameter a scenario-based calculation has to be performed (e.g. assumption of a constant worst case wind speed during the application based on an evaluation of weather data from the local station). However, these artificial calculations could hardly be validated.

Also the estimation of wind speed based on regional wind speed and slope as it is done by wind energy producers would not solve the principal problem, because their models are suitable to calculate wind velocities at heights above grounds similar to the wind power stations and not close to the surface (see Figure 4-1).

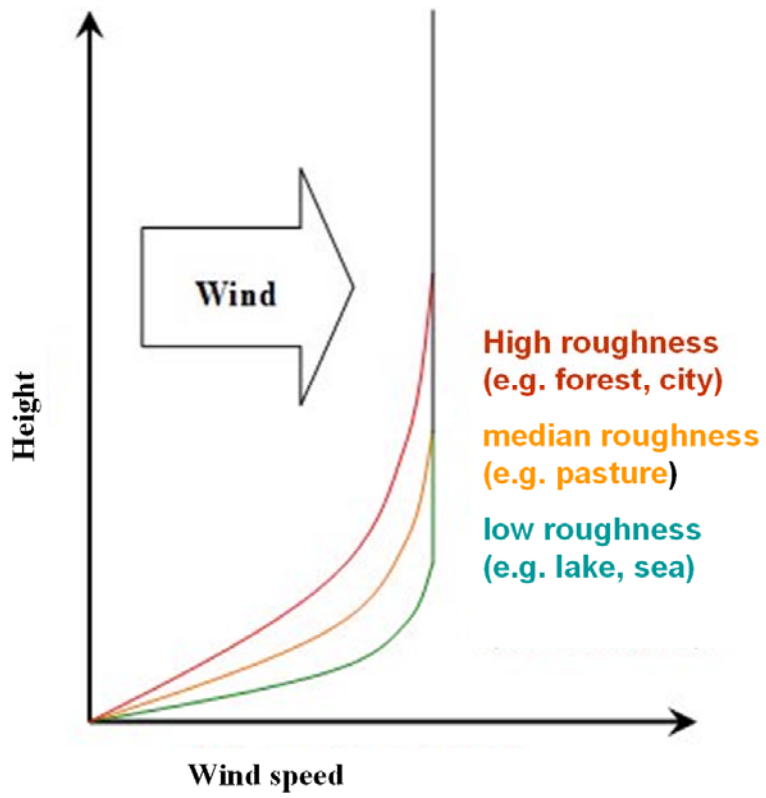

Figure 4-1: Dependency of wind speed on height above the surface (Bundesverband Windenergie 2010)

4.2.1.3 Sensitivity of wind speed

Based on the simple box model explained earlier the following estimation of the sensitivity of wind speed on the expected exposure into surface water bodies can be given:

Figure 4-2 shows calculated drift percentiles dependent on wind speed and travel distance for droplets of 100 µm diameter (deposition velocity: 25 cm/s) based on the box model described earlier.

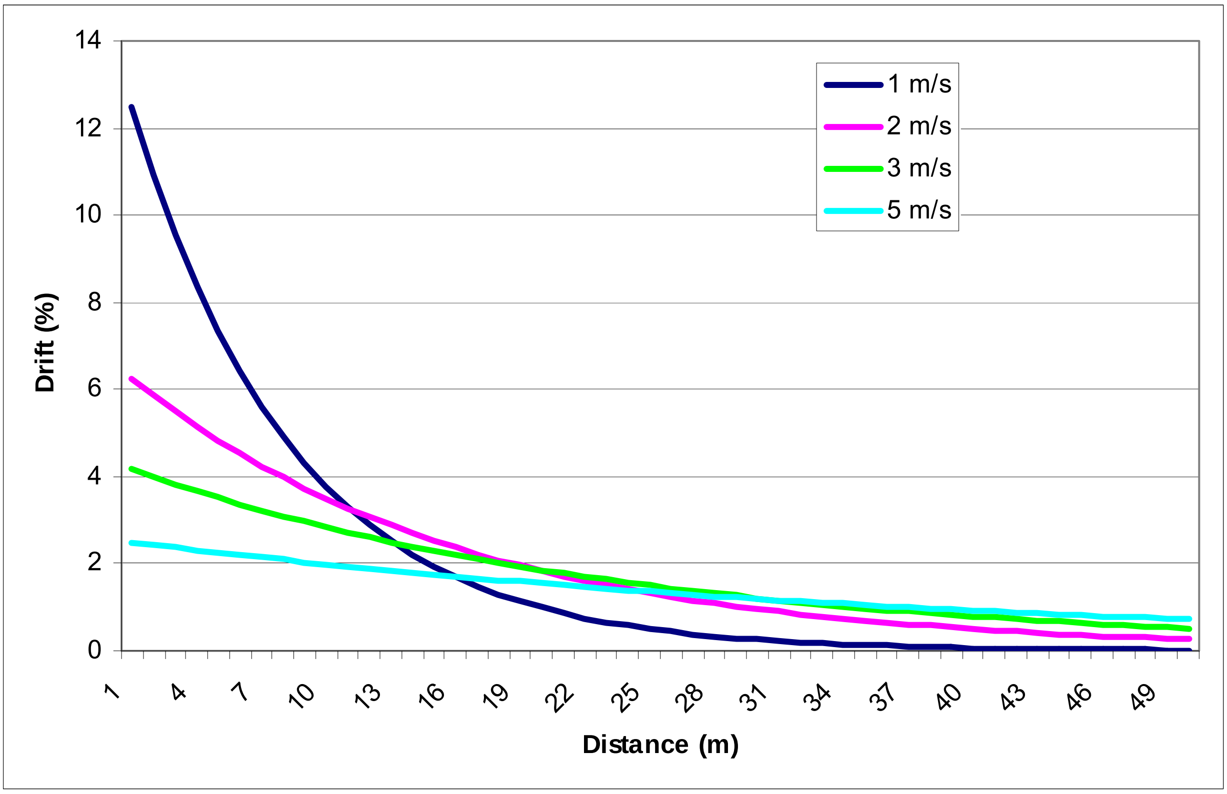

Figure 4-2: Calculated drift input per unit area (percent of the application rate) based on wind speed and travel distance

Obviously, the absolute pesticide amount leaving the field via spray drift increases with increasing wind speed. This is the background for the prohibition of spraying at high wind speeds.

On the other hand, the reduction of the decline per distance unit in the equation leads to smaller entries very close to the treated area (and consequently smaller initial concentrations in the surface water body). With increasing distance from the field this effect reverses and leads to higher loadings compared to small wind speeds.

It can be concluded that high wind speeds do not necessarily lead to high initial concentrations, but lead to a significant increase of the overall transport of pesticides out of the neighbouring area.

4.2.1.4 Deposition without wind

According to the theoretical approach of the simple box model (Figure 4-2) very low wind speeds of less than 1 m/s in the absolute short-distance-range below 2 m could lead to higher deposition. Within this work package literature and drift deposition data were analysed in order to crosscheck this assumption. In addition, experts have been interviewed on this subject.

The outcome forms the basis for deciding on whether or not deposition without wind should be considered explicitly in the new approach.

Wind is assumed to be the most important factor for the aerial transport of droplets beyond field boundaries (Göhrlich 1982 and Koch 1989 in Koch et al. 2005). Looking at the absolute short-distance-range low wind speeds increase the influence of random impacts on the drift process (Kaul 2009 and Koch 2009).

For high wind speeds the conclusion drawn from the simple box model (Figure 4-2) corresponds to Wang and Rautmann (2008) who found that wind speed was correlated with spray drift. But a significant correlation was only detected at distances of 15 m or more from the treated fields. Von der Hude (2004) found an influence of wind speed only in the earlier field trials (Ganzelmeier et al. 1995). This influence could not be confirmed for the later trials of Rautmann et al. (2001). The analyses of the authors mentioned above were based on the JKI drift datasets for field crops. None of the authors analyzed drift deposition at very low wind speeds in the absolute short-distance-range explicitly.

Table 4-3: Limits of determining parameters for the calculations with PeDriMo (Kaul et al., 2004)

Parameter		**Limits**
Gaps in the foliage	%	30...85
Nozzle height above tree	cm	245...280
Difference nozzle direction and tree	cm	20...50
Air volume	m³/h	20000...44000
Liquid volume	l/ha/m	160...200
Type	-	1...5
Wind speed	m/s	2…5
MVD	µm	200...300
Air temperature	°C	11...25
Psychrometric difference	K	3...6
Hours since sun start	hours	6...13
Numbers of rows with both side-application	rows	3...5
Vehicle speed	km/h	6

Also the mathematical/physical model “PeDriMo” (Pesticide Drift Model) which is based on 129 trials with 8944 measurements (Kaul et al. 2004) is limited to wind speeds > 2 m/s (see Table 4-3). According to Kaul (2009) this model limitation is due to the limited number of drift measurements at wind speeds below 2 m/s (see Table 4-3).

However, statistical tests for hypothesis testing were performed for the available drift deposition data for hops, orchard and field crops using the statistical software program SAS 9.1 (SAS Institute, 2002) to find out whether there are higher drift depositions at low wind speeds. Differences between means were analysed using t-tests. The individual measurements of the trials are grouped per measurement distance (xmin) into the classes: Higher wind speed (>=1.5 ms-1) and Lower wind speed (<1.5 ms-1). The numbers of the trials per measurement distance and wind speed group as well as the means (mean_x) and the standard deviation (std_x) of the trials are illustrated in the appendix.

The available data for analysing deposition for low wind speeds is limited. Although the means of the deposition in hops and field crops are higher at low wind speeds the results are not statistically significant. We find a statistical difference in the orchard deposition data. Depositions at high wind speeds are higher compared to low wind speeds for all distances. For the statistical values of the deposition trial data for orchards and hops see Appendix of chapter 4.

4.2.1.5 Conclusions: Parameter wind speed

Literature and experts state that an exact calculation of drift entries based on explicit wind speed is difficult due to the extreme variations in time and space during the application. The drift deposition data used in national and EU risk assessments stem from field studies in Germany (Ganzelmeier et al. 1995, Rautmann et al. 2001). They have been carried out under conditions of good agricultural practice. The trials were conducted at various sites on different days and therefore inherit different realistic application conditions. The drift conditions during the trials are documented and published in Ganzelmeier et al. (1995) and Von der Hude (2004). A limited number of trials at low wind speeds is included. There is not state-wide data available for wind speed at relevant heights above ground (~2 m). At a driving speed of 6 km/h a wind movement of 1.6 m/s (6000 m/3600 s) in driving direction is caused by the sprayer. This speed should not be exceeded in vine and fruit growing according to GAP (Bundesanzeiger 2010)[1]. The available field trial data show no statistical difference between the trials per measurement distance and wind speed group. The results do not support the assumption that drift deposition is underestimated at low wind speeds.

It is therefore recommended not to consider wind speed in the new assessment explicitly. But it is recommended to consider the available drift deposition data in the new approach in a way that ensures the “between trials” variability due to different realistic application conditions.

Last, but not least also the existing drift percentiles are generally accepted and widely used in European pesticide registration. A correction of these drift values does not seem reasonable in the present situation.

4.2.2 Parameter wind direction

The wind direction is not considered in the current procedure because it is always assumed that the wind is directed to the surface water (+/- 30° (BBA, 1992)) (worst case condition). If the wind direction was considered within the registration procedure usually smaller concentrations in the surface water would be simulated for the individual directions.

In landscape level risk assessment the number of wind directions is limited to eight main directions (N, NE, E, SE, S, SW, W, NW) (see Hendley et al. 2001b; Schad et al. 2006; Urban 2003; Holmes et al. 2007). The PEC_{ini} distributions for each of these eight pre-determined directions theoretically consider the spray drift coming from a wind window of 45° (+/- 22.5° to the entry direction, see Figure 4-3). The window of 60° (+/- 30°) according to the BBA method for measuring direct drift in the field (BBA, 1992) is not exceeded and therefore the approach of limiting wind directions to eight goes in line with BBA (1992).

[1] Bundesanzeiger 2010: Bekanntmachung der Grundsätze für die Durchführung der guten fachlichen Praxis im Pflanzenschutz, Jg. 61, Nr. 76a, 21.5.2010

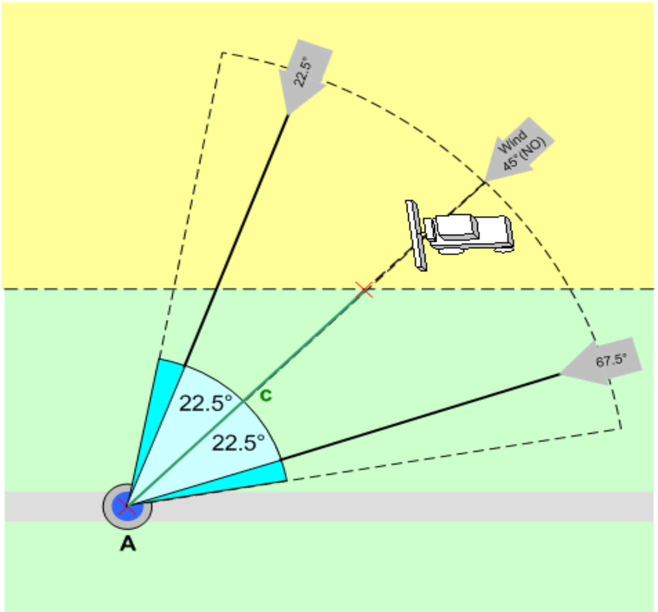

Figure 4-3: Entry direction 45° with the theoretical drift window of 22.5° (solid line) and the theoretical drift window according to BBA (1992) (dashed line)

4.2.2.1 Calculation of spray drift considering the wind direction

It is not complicated to consider the wind direction in the geo-referenced software because all necessary geometric information is available directly in the system. However, dependent on the wind direction different water body segments may be affected at different distances leading to different loadings.

Generally, for the calculation of entries into surface water segments the wind direction could be considered in different ways:

- "to perform PEC calculation based on the true wind direction (measured) for each adjacent application side (realistic case, but theoretic),"
- "to perform PEC calculation based on the true wind direction which results in the shortest distance between the application side and the surface water segment (worst case),"
- "to perform PEC calculation for each of the 8 main wind directions…"
  (Hendley et al. 2001a; Golla et al. 2007; Schad 2007; Holmes et al. 2007).

4.2.2.2 Availability of the input parameter wind direction

Similar to the wind speed the necessary information is not available at the field scale. Generally, weather data could be made available from weather stations in the neighbourhood. However, as the wind direction is an extremely dynamic factor with fluctuation even during the experiment only a scenario-based consideration would be possible. The high fluctuation of the wind direction on the field scale was also an experience of the experimental groups who carried out the drift experiments. Because of the high fluctuation of the wind direction it was difficult to carry out the experiments because they could only be performed if the wind was in direction of the petri dishes (Kubiak, 2009).

Due to these characteristics it is only possible to consider the wind direction based on the second (worst case) and third option.

4.2.2.3 Sensitivity of the parameter wind direction

It can be expected that on the local scale (water body segment) the loadings are significantly influenced by the wind direction. Figure 4-4 illustrates that the distance between water body and application device depends on the spatial orientation and the wind direction. In both examples the shortest distance will be for a wind blowing directly from the north or south direction. According to the regression function of Rautmann (JKI, 2008) for orchard (early stage) and single application (90th percentile drift) the sediment [%] varies depending on the wind direction: 29.9 (3 m), 27.8 (3.2 m), 22.7 (4.2 m), 14.2 (7.8 m).

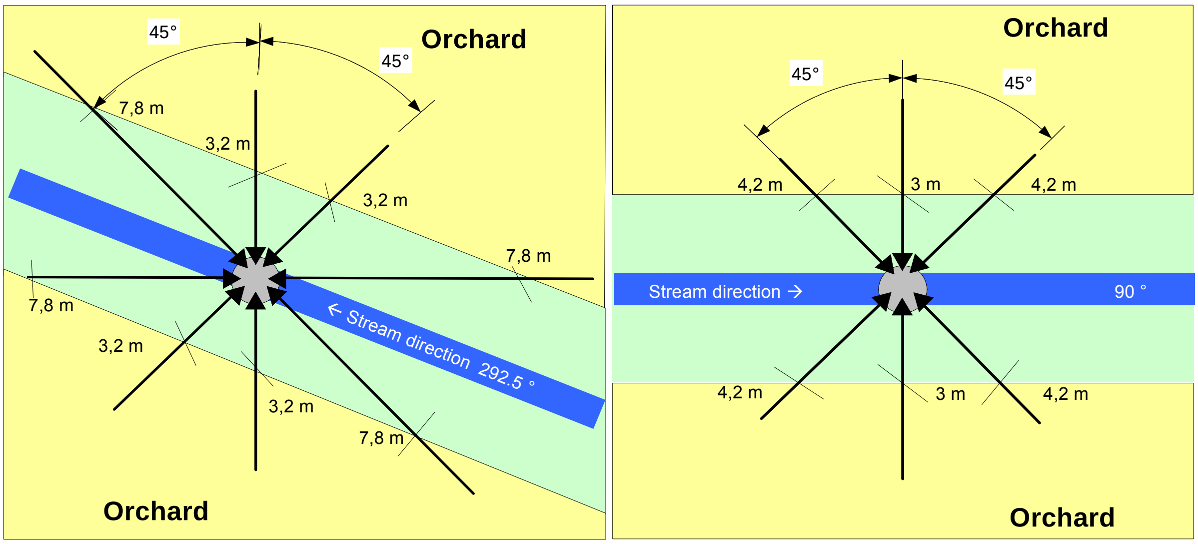

Figure 4-4: Distance to water body depending on wind direction for two different stream orientations

But already at the regional scale the integrated loadings will hardly be influenced due to averaging effects.

In order to clarify the question on how many wind directions may be necessary to sufficiently describe possible local entries into the surface water and how parameter settings affect the total length of the calculated hotspots (here called risk management segments, *RMS*) a sensitivity analysis was performed.

High resolution geodata from the hop growing region “Hallertau” (see Figure 4-5) was selected for this analysis. In each case a 1.795 km total length of water bodies was examined. The stream network was divided into 183273 segments (10 m length). Since there were sections smaller than 10 m during the segmentation, the exact number of water segments in the area was found to be 179500 (see also Wagner et al. 2007; Trapp et al. 2008).

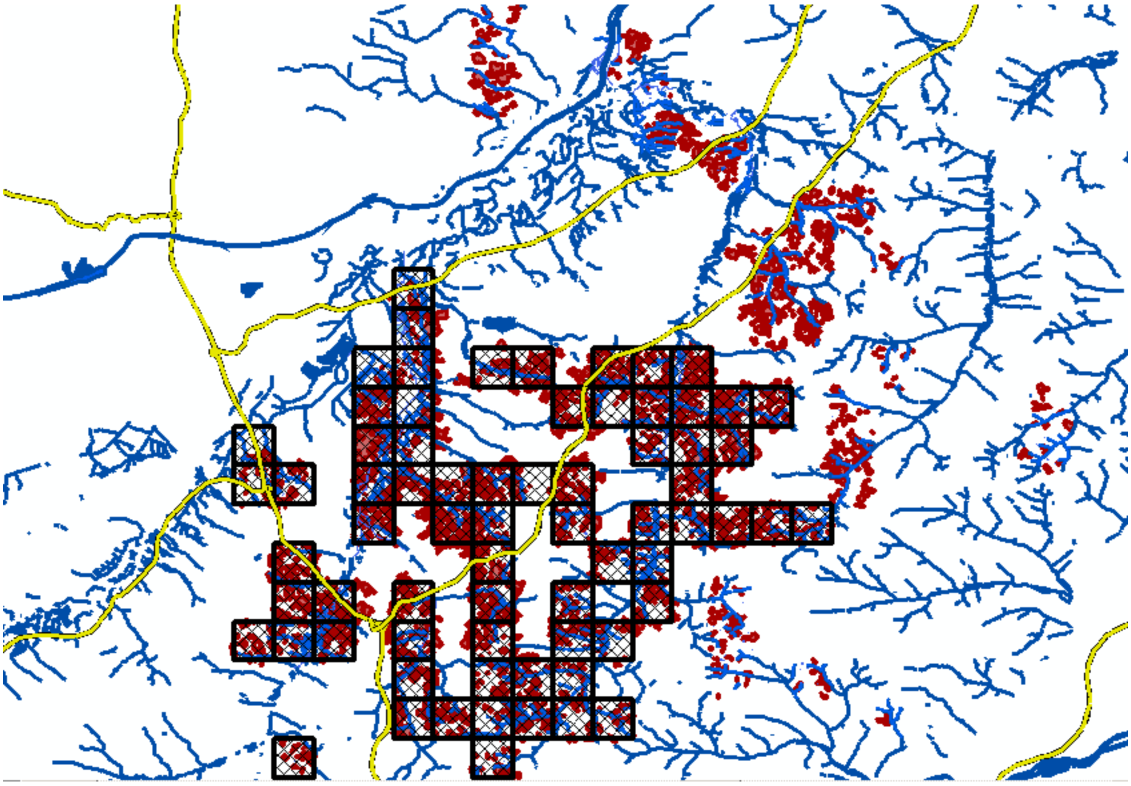

Figure 4-5: Overview Region Hallertau: hops and water bodies based on the ATKIS DLM2 and the hr-analysed region as black hatched grids (Presented: Deutsche Pflanzenschutztagung 2006, Göttingen)

“Moving window”-method

The **“moving window”-method** (UBA 2007, not published) developed as a preliminary hot spot criterion which accounts for potential re-colonisation and recovery processes and does therefore allow to identify ecologically critical spatial accumulation of segments with PEC >RAC, uses three parameters:

- the length of the temporary “moving window” (default 1000 m)
- the percentage of segments with PEC>RAC within the defined moving window (default 10 %) and
- the height of the maximum exceedance of one segment of the RAC (default 10 times).

Procedure for calculating the PEC

Based on the drift entries resulting from the calculated wind direction different procedures can be used to calculate the amount finally reaching the surface water.

For the following analysis always the mean value of the calculated wind directions was used based on the methodology described in the GeoPERA-project. This method generally differs from the approach developed in the GeoRisk-project. Nevertheless, the different methods have to lead to comparable results due to the fact, that only the number of wind direction was modified but the method to calculate the resulting PEC-values did not change (mean value of the calculated directions).

Assumptions:

- PEC-calculation
- Equal distribution of wind directions (PEC_{mean})
- Standardised water body (300 litres)
- 90^{th} percentile of the Rautmann values (fitted regression curve)
- Using a combination of high resolution landscape classification data and the “Objektart” 4109, permanent crops, hops from the ATKIS DLM2-data
- The RMS-calculation (hotspot-calculation) was conducted following the concepts of the moving window method (UBA 2007 not published) (RAC exceedence in not more than 10 % of the segments along 1000 m water body).
- The RMS (hotspot)-identification was based on the foregoing calculation of the PEC_{mean} for every water segment (10 m long).

In the following, analysis calculations were done considering 4, 6, 8, 12 and 16 wind directions.

To get a more general feeling about the dependencies additionally different buffer sizes around water bodies for the simulation of buffer strip requirements (3 m and 10 m) were used (see Table 4-4).

Table 4-4: Overview of the parameter variations considered in the analysis

Number of wind directions	4, 6, 8, 12 and 16
Buffer size around water bodies:	3 m and 10 m
Length of the Moving-Window:	1000 m, 2000 m, 3000 m, 4000 m
Resulting minimal RMS-length:	100 m und 200 m (through different combinations of Moving-Window-length and segment percentage)

The following four tables show the results of the analysis for different situations. Reference base is the calculation with 8 directions; thereof the percent difference is specified.

Table 4-5: Results of the analysis for a moving-Window of 1000 m and 3 m-buffer:

Number of wind directions	4	6	8	12	16
Number of RMS-segments	6814	6975	6979	6945	6928
Length [km]	67.6	69.18	69.21	68.86	68.68

Table 4-6: Results of the analysis for a moving-Window of 1000 m and 10 m-buffer:

Number of wind directions	4	6	8	12	16
Number of RMS-segments	2555	1942	1983	1966	n.c.[1]
Length [km]	25.37	19.23	19.68	19.49	n.c.[1]

[1] no calculation

Table 4-7: Results of the analysis for a moving-Window of 2000 m and 3 m-buffer:

Number of wind directions	4	6	8	12	16
Number of RMS-segments	6289	6453	6463	6408	6419
Length [km]	62.38	64	64.09	63.53	63.64

Table 4-8: Results of the analysis for a moving-Window of 2000 m and 10 m-buffer:

Number of wind directions	4	6	8	12	16
Number of RMS-segments	2045	1577	1631	1572	n.c.[1]
Length [km]	20.3	15.6	16.17	15.57	n.c.[1]

[1] no calculation

In the following two figures some detailed results are further highlighted.

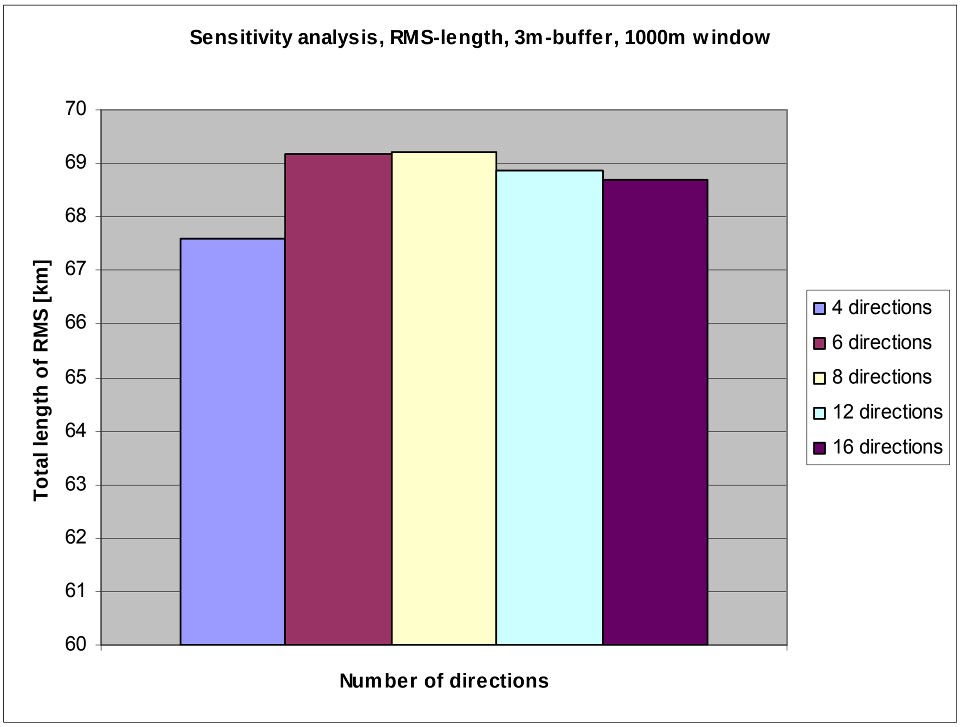

Figure 4-6: Hotspot (Risk Managament Segment, RMS) length dependent on the number of wind directions (3 m buffer, 1000 m window)

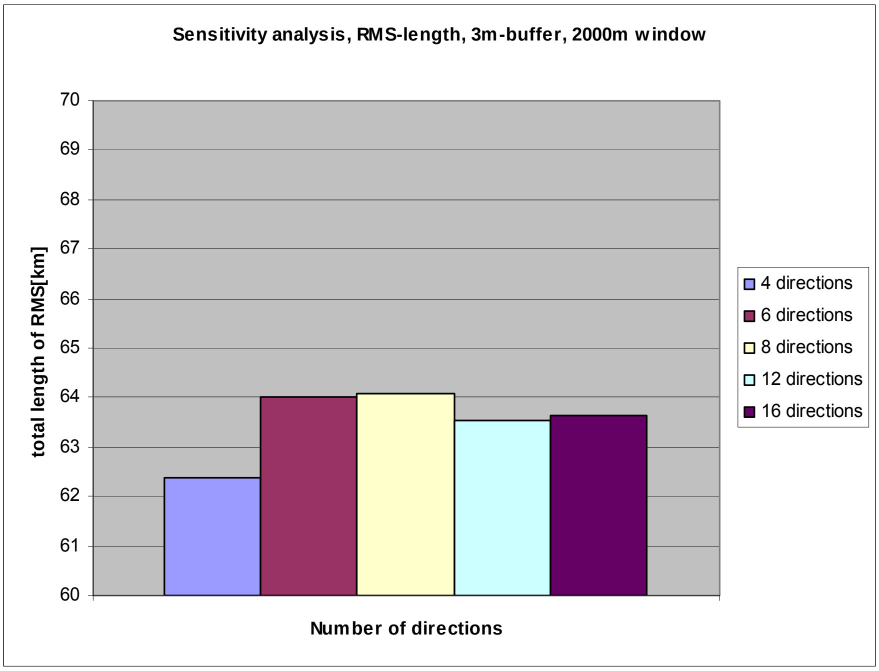

Figure 4-7: Hotspot (RMS) length dependent on the number of wind directions (3 m buffer, 2000 m window)

Based on the analysis with hr-geo-data performed for the hop growing region Hallertau following conclusions can be drawn regarding the essential number of wind directions to be considered and the effect of different minimum buffer sizes:

- Already at a simulated buffer strip requirement of 3 m there is only little influence of the number of wind directions on the total length of hotspots (RMS) (difference of max. 3 %).
- At a simulated buffer strip requirement of 10 m the total length of hotspots (RMS) does not vary regardless of whether 6, 8 and 12 wind directions have been considered (variation max. 4 %). However, if only 4 wind directions are taken into account clear differences in the results (around 26 % difference with Moving-Window-length of 1000 m and 29 % difference with a Moving–Window of 2000 m) were found. A number of 8 wind directions is necessary to guarantee stable results in the analysis.
- An enlargement of the buffer size (simulated buffer strip requirement) leads to a reduction of the hotspots (RMS).

Based on the same analysis with hr-geo-data (Hallertau) the following conclusions can be drawn for the effects of the moving window length and the critical segment percentage with PEC > RAC on the total length of the RMS:

- An enlargement of the Moving-Window leads to a reduction of the hotspots (RMS) (if the segment percentage leading to a RMS designation is left constant).
- An enlargement of the resulting minimal RMS-length through an increasing segment percentage leads to a reduction of the total RMS-length.
- The segment percentage has a stronger influence on the total hotspot-length (RMS-length) than the size of the Moving-Window.

During a comprehensive "ground truthing" in the hop region Hallertau (Trapp et al. 2009; Trapp et al. 2008b), additionally the waterside parts of hop fields (length, width) were randomly measured. From this, it can be considered that, in general, the waterside extent (length, width) of the hop fields often exceeds 100 m length. Therefore the length of segments with PEC > RAC exceeds the length of 100 m, too. Hence, the effects of the “moving window” method are low, because most of the segments with PEC > RAC have lengths about 100 m and are classified automatically as a hotspot.

77.90 km single segments with PEC > RAC were calculated including a buffer of 3 m using the methods described by Trapp et al. (2008). Using the moving window method described by UBA (2007 (not published)), 69.21 km RMS were calculated. This means that the effect of the moving window-method was a reduction of 8.69 km length (11 %).

Regarding the effect of a 10 m buffer, 24.10 km single segments with PEC > RAC were calculated using the methods described by Trapp et al. (2008). Using the moving window method described by UBA (2007 (not published)), 19.68 km RMS were calculated. This means that the effect of the moving window-method was a reduction of 4.42 km.

The small effect of the applicaton of the UBA hotspot criterion probably results from the specific properties of the Hallertau: If there are hop areas close to a stream then these are usually found along more then 100 m.

4.2.2.4 Conclusions: wind direction

As the exact direction of entry can hardly be predicted it follows the idea of a probabilistic approach to consider a number of wind directions as possible entry directions. The GIS-analysis showed that 8 wind directions are a reasonable number for calculating the entry intro surface water, and the results do not change if more than 8 directions are considered.

4.2.3 Parameter nozzle technique

Already for a long time the nozzle technique has been considered by correcting the standard drift percentiles with constant reduction factors. Without this risk management measure some of the pesticides on the market could hardly be registered due to their toxic potential.

4.2.3.1 Calculation of drift entries considering nozzle technique

The calculation of drift entries depending on nozzle type is rather simple, because constant reduction factors dependent on the nozzle types are available. The same equation could be easily transferred to a geo-referenced system without additional modifications.

$D_{red} = D_0 * d_{red}$

d_{red}: drift reduction because of advanced nozzle technique (-)

D_0: standard drift percentile with classic technique (%)

D_{red}: reduced drift percentile for advanced nozzles (%)

4.2.3.2 Availability of the drift reduction factors

Presently reduction rates from 50 % up to 99 % are possible dependent on spraying equipment. However, high reduction rates can only be achieved when using special nozzles. According the the German association of hop farmers for hop reduction rates up to 99 % are achievelable. Nozzles are classified with respect to their reduction rates and it is guaranteed that the necessary reduction is achieved independent on the distance from the field. The same factors could therefore be used in the new probabilistic system.

4.2.3.3 Sensitivity of the drift reduction factors

Dependent on the actual nozzle type the initial concentration can be influenced by more than one order of magnitude (avaiable reduction factors 50 %, 75 %, 90 %, 99 %). In the past many pesticides received registration only after considering drift reducing nozzles during application. As the parameter is not geo-referenced, changing the nozzle type will only influence the initial load. Neither the spatial distribution nor the time dependency of concentration in the surface water will be affected.

4.2.3.4 Conclusions: nozzle technique

The drift reduction by special nozzle types is an important factor to reduce possbile concentrations in surface water close to the field. It is recommended to consider drift reducing factors in the new system as well.

As no spatial distribution of this parameter has to be considered it could be easily implemented into the new system.

It is therefore recommended to use the same reduction factors in the new system as in the current assessment scheme.

4.2.4 Parameter spray drift deposition rate

The spray drift deposition rate [%] due to direct drift can be calculated using a deterministic or probabilistic approach:

With a deterministic approach the spray drift deposition is calculated using the regression formula of the 90^{th}-percentile drift values (for a single application), as described by Rautmann (2001):

$F(x) = a * x^b$

F(x) = drift deposition rate [%]
x: geo-referenced distance in a given wind direction
a: crop dependent fitting parameter (Rautmann 2001)
b: crop dependent fitting parameter (Rautmann 2001)

Another deterministic approach is the drift calculation according to FOCUS (2002). In FOCUS 2002 the same experimental data set has been used. But additionally to single applications drift rates are given for multiple applications as well. For multiple applications the drift values represent overall 90^{th} percentiles with lower drift percentiles for the individual spraying events (e.g. two applications per season: 82^{nd} percentile instead of 2* 90^{th} percentile). This may lead to smaller initial concentrations for multiple applications compared to single applications if the pesticide is significantly degraded or transported during the time between two applications. In order to ensure a worst case situation the PECs resulting from different applications can be simply added without considering degradation/transportation.

To consider the variability of spray drift deposition resulting from different realistic application conditions a probabilistic approach is appropriate. The main idea is that in an exposure assessment a probabilistic approach should take into account the variation between spray events instead of just using e.g. the 90^{th}-percentile drift rate of the individual measurements. Using this approach the effect of all processes and factors leading to the between-trial variation are indirectly considered, even those which are random.

A probabilistic approach is conducted within the EUFRAM Project for a single distance of 1 m in field crops (EUFRAM 2006). The underlying idea is that using all the individual measurements in the drift deposition dataset (variation between the 1 m drift deposition collectors) to form a distribution for spray drift may not be appropriate in a probabilistic procedure. Schafer et al. (2005) consider a trial-by-trial analysis using the average values from each trial more suitable for a probabilistic approach. They state that one spraying event is the appropriate statistical population. A trial-by-trial analysis reflects the variation between spraying events due to different application conditions (wind speeds, wind direction, nozzle, vehicle speed, relative humidity, etc.).

Wang and Rautmann (2008) propose a probabilistic method which uses drift deposition values only for these measurement distances of the trials. Drift depositions are not computed for any geo-referenced distance.

Golla et al. (2008) propose an approach for modelling spray drift deposition distributions for any distances up to 75 m (low crops) and 150 m (high crops).

The approach follows the idea of a trial-by-trial analysis of drift measurement datasets applied within the EUFRAM project for arable crops and considering only a single distance (EUFRAM 2006). In a probabilistic assessment a trial wise analysis is more appropriate. It allows for considering the variation of application conditions between trials such as air temperature, wind speed, wind direction, nozzle, vehicle speed and rel. humidity.

At random appearing influences (turbidity, spontaneous alteration in wind circumstances) affect the small scale variability of in between petri dishes (1 m). In a PEA this small scale variability is considered as non-characteristic for an application event (one experiment) (EUFRAM 2006).

According to the code of practice the meteorological and procedural conditions (wind speed and direction, temperature, nozzles, vegetation etc.) for an experiment (10 m measurement track) can be assumed as nearly constant.

In a spatial PEA the transfer of phenomena of individual petri dish measurements to a water body segment with a length of e.g. 25 m is not reasonable, since an extremely high or low drift in a petri dish is not considered to be a good indicator of the deposition rate to which one water body segment might be exposed.

The approach of an experimental evaluation of drift measurements renders the discrete observance of wind strength unnecessary, since this influence is expressed in the variation of the drift experiments.

Individual regression curves were developed for four groups (arable crops, grapevine, fruit crops and hops). The general approach is described for fruit crops. For each crop type the individual drift measurement datasets of Ganzelmeier et al. (1995) and Rautmann et al. (1999) are analysed on a trial-by-trial basis.

In a first step the mean of the deposition rates per trial are computed and distributions of the trial means and the single measurements are compared.

The trial means can be described as a lognormal distribution. For each of the trials and the measurement distance x the logarithm of the trial mean is computed. For each measurement distance x the logarithmised trial mean can be plotted as normal distribution, which again can be described with a mean and a standard deviation.

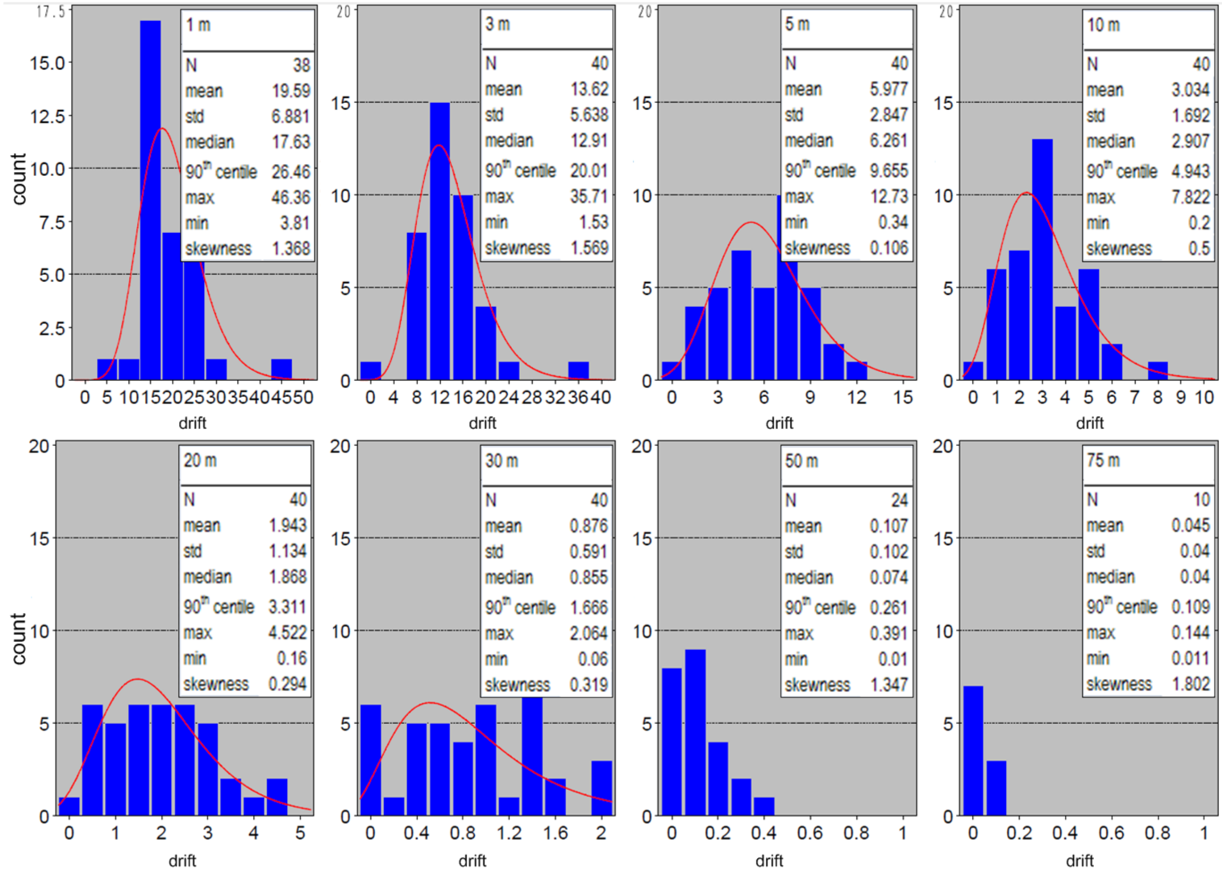

Figure 4-8: Frequency distribution of the trial means of drift deposition rates for different distances from fruit crops (orchard early stage)

Functions (1) and (2) are fitted using non linear regression (PROC NLIN; SAS Institute 2002) for the logarithmised trial mean and linear regressions for the logarithmised standard deviations as shown in Figure 4-8. The regression lines for permanent crops and the vegetation development stages are displayed in Figure 4-9 to Figure 4-12.

$$m\_lndrift(y) = a\ \ln(y)^2 + b\ \ln(y) + c \qquad (1)$$

$$s\_lndrift(y) = a\ ln(y) + b \qquad (2)$$

where m_lndrift and s_lndrift describe the mean and the standard deviation of a deposition value as a function of distance y

Putting m_lndrift(y) and s_lndrift(y) in a function for random normal distributions (RAND; ditto) a logarithmised deposition value lndrift is computed (3).

lndrift(y) = rand(norm;m_lndrift(y); s_lndrift(y)) (3)

The modelled spray drift deposition value follows equation (4):

drift = exp(lndrift(y)) (4)

Table 4-9: Model parameters (A, B, C, D*) and hinge distance [m]

crop groups	**Function**	**A**	**B**	**C**	**D**	**Hinge distance [m]**
Pome fruit early	(1) mean	-0.4346	0.3967	2.9852		
Pome fruit early	(2) std	0.2175	0.1665			
Pome fruit late	(1) mean	-0.2994	-0.2672	2.5292		
Pome fruit late	(2) std	0.2163	0.3871			
Hops	(1) mean	-0.4062	0.0698	2.7463		
Hops	(2) std *	0.3549	0.0000	-0.4584	2.5214	22.205
Vines	(1) mean	-0.2747	-0.4235	2.3197		
Vines	(2) std	0.0284	0.305			

The model has been tested against the available experimental data and their distribution. The 90th percentile of the output distribution is compared to the 90th percentile of the trial means and 90th percentile of the single measurement values. Table 4-10 is an example for the test of the model for orchard early trials. The agreement affirms that with the described method the real drift distribution can be imaged for any distances.

Table 4-10: Comparison of the 90th percentile of the drift measurement data in orchards:

Distance (m)	3	5	10	15	20	30	50	75
	N=351	*N=453*	*N=453*	*N=448*	*N=413*	*N=413*	*N=253*	*N=100*
Individual results	27.33	23.48	10.63	5.996	3.744	1.740	0.268	0.098
	N=38	N=40	N=40	N=40	N=40	N=40	N=24	N=10
Mean results	26.46	20.01	9.655	4.943	3.311	1.666	0.261	0.109
Calculated values n=10000	30.44	23.54	11.59	6.31	3.75	1.60	0.45	0.46

Row 3: Analysis of individual measurements (Ganzelmeier et al. 1995; Rautmann et al. 1999);
Row 5: Analysis of the trial means;
Row 6: Simulation of spray drift deposition according to the proposed approach

At random appearing influences (turbidity, spontaneous alteration in wind circumstances) affect the small scale variability of in-between petri dishes (1 m). In a PEA this small scale variability is considered as non-characteristic for an application event (one experiment) (EUFRAM 2006).

According to the code of practice the meteorological and procedural conditions (wind speed and direction, temperature, nozzles, vegetation etc.) for an experiment (10 m measurement track) can be assumed as nearly constant.

In a spatial PEA the transfer of phenomena of individual petri dish measurments to a water body segment with a length of e.g. 25 m is not reasonable as an extremely high or low drift in a petri dish is not considered to be a good indicator of the deposition rate to which one water body segment might be exposed.

The approach of experimental evaluation of drift measurements renders the discrete ob-servance of wind strength unnecessary, since this influence is expressed in the variation of the drift experiments.

a) b)

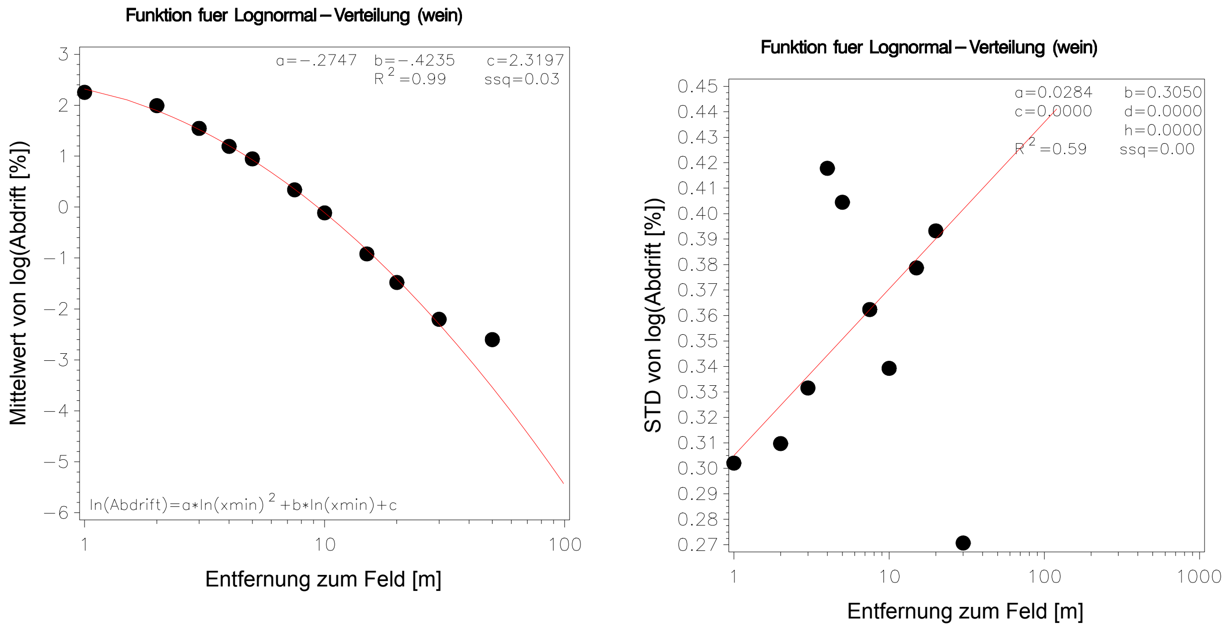

Figure 4-9: Grapes: Regression line to the mean of the logarithmic trial means (a) and the logarithmic standard deviation (b) at measurement distance x

a) b)

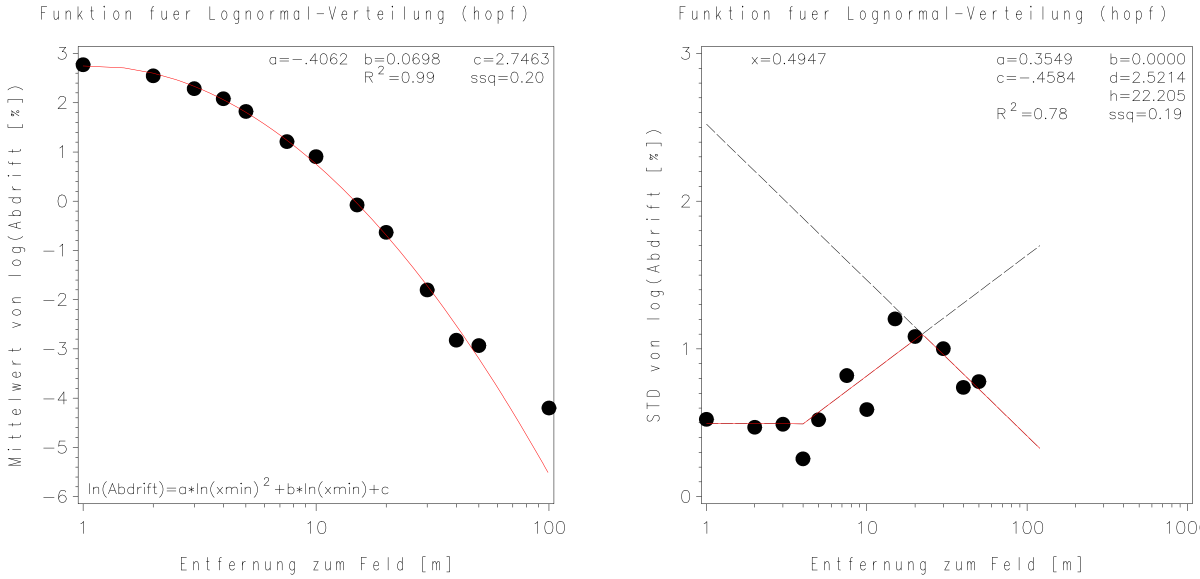

Figure 4-10: Hops: Regression line to the mean of the logarithmic trial means (a) and the logarithmic standard deviation (b) at measurement distance x

a) b)

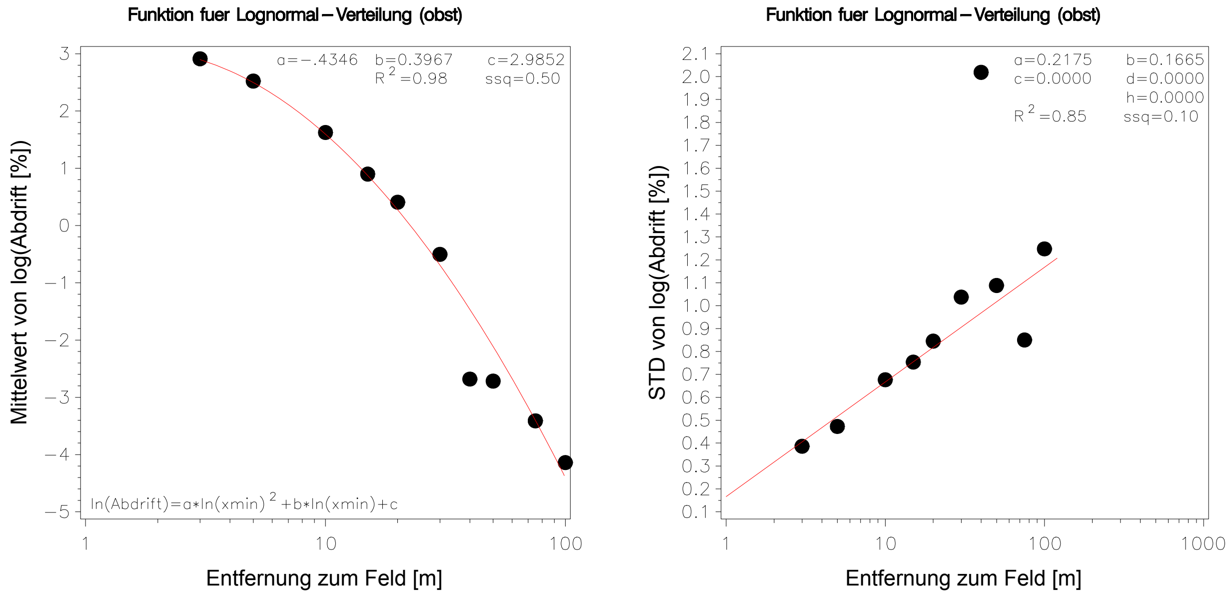

Figure 4-11: Orchard early stage: Regression line to the mean of the logarithmic trial means (a) and the logarithmic standard deviation (b) at measurement distance x

a) b)

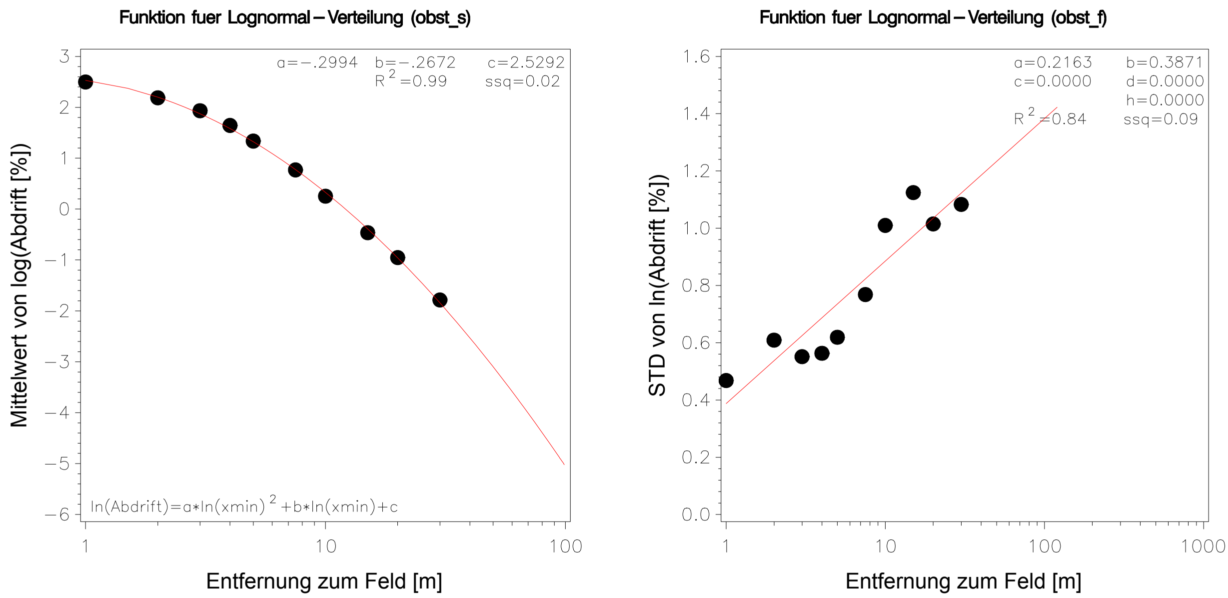

Figure 4-12: Orchards late stage: Regression line to the mean of the logarithmized trial means (a) and the logarithmized standard deviation (b) at measurement distance x

4.2.4.1 Availability of the input parameters

Underlying data has been published and is electronically available (spray drift trial data of Ganzelmeier et al. 1995 and Rautmann et al. 1999). The collection contains the wind speed variation with drift trials. It can be expected that wind speed varies also within drift trials. However, the simulation of these variations is not covered by the simple box model. These additional variations could only be estimated based on a new evaluation of the original experimental data based on the individual measurements rather than the mean.

4.2.4.2 Conclusions: drift deposition rate

It is recommended to calculate spray drift distributions for each of the wind directions of a water body segment based on the trial means and their standard deviation, as this is also recommended in (EUFRAM, 2006). The approach takes into account the variability of drift deposition rates due to “between trials” variability, which also expresses the variability in the trial application conditions (e.g. wind speed, wind direction, nozzle, rel. humidity etc). To calculate a probabilistic drift deposition rate according to the idea of EUFRAM for any geo-referenced distance functions for different crop types are proposed (Golla et al. 2008, 2009).

4.2.5 Parameter waterside vegetation

Currently a reduction of the spray drift loading due to vegetation between the field and the surface water is not considered in the current (worst case) scenario. However, it is expected that this vegetation has a significant effect on the drift entry.

4.2.5.1 Calculation of drift entries considering water side vegetation

The calculation of drift entries dependent on shielding vegetation could be done rather simply, if constant reduction factors were available based on the vegetation type or the leaf stage. The equation could also be used in geo-referenced systems without additional modifications.

$D_{red} = D_0 * d_{red}$

d_{red}: drift reduction due to shielding waterside vegetation (-)

D_0: standard drift percentile with classic technology (%)

D_{red}: reduced drift percentile if waterside vegetation is present (%)

The consideration of the waterside vegetation will only influence the initial loading but does not affect the time dependency of concentrations in surface water.

4.2.5.2 Availability of the input parameters

For the consideration of waterside vegetation in the geo-referenced model two factors must be available, firstly, information of the existence of waterside vegetation in the system and secondly, drift reduction factors as a function of vegetation type and/or the leaf stage.

An inventory of observed reduction factors caused by water side vegetation has been generatied by Schulz et al. (2007) which shows efficiencies up to 90 %. A summary of these drift factors is given in Table 4-11. Reduction factors are mainly dependent on the leaf stage with minimum reduction factors of about 10 % to 30 % at early stages and about 90 % for the fully developed vegetation. Experimental results demonstrate that the effectiveness of vegetation as a shield depends on its height and porosity (Ucar and Hall 2002). Generally, the height of vegetation serving as a shield should be twice as high as the target crop. Very dense vegetation is not as effective as porous vegetation, presumably because the drift cloud is redirected upwards rather than absorbed (Wolf and Cessna 2006; Davis et al. 1992; Ucar and Hall 2001). Compared to the height and the porosity the width of the vegetation is less important (Ucar and Hall 2002).

Table 4-11: Spray drift reduction factors for different types of vegetation*

Type	Leaf stage	Remarks	Drift reduction (%)	Reference
Windbreak	early		67.5	Schad 2006
Windbreak	early		68 - 79	Van de Zande et al. 2005
Windbreak	early l		68-90	Ucar & Hall 2001
Windbreak	early	alder	50	Richardson et al. 2004
Windbreak	early	alder, height: 4-5 m	10-30	Wenneker et al 2005
Windbreak	fully developed	edge of field	70-90	Ucar & Hall 2001 Hewitt 2001
Windbreak	fully developed	salix und casuarina, height: 8-10m	98	Ucar & Hall 2001
Windbreak	fully developed	shelter vegetation	75-88	Schad 2006
Windbreak	fully developed		up to 90	Van de Zande et al. 2005 Hewitt 2001
Windbreak	fully developed	alder, height: 7 m e	86-91	Walklate 2001
Windbreak	early	alder, height: 4-5 m	63-85	Wenneker et al 2005
Hedge	-	alder	70-85	Richardson et al. 2002
Windbreak	-	alder	80	Richardson et al. 2004
Windbreak	-	alder	70-90	Schad 2006
Hedge	-	width: 5 m	73 %	Drew 2005
Hedge	-	width: 5 m	91 %	Drew 2005
Hedge	-	width: 1.2 m height: 1.6 m	65 %	Davis et al. 1992
Hedges, windbreak	no leaves	evaluation of literature	25 %	FOCUS 2004
Hedges, windbreak	medium	evaluation of literature	50 %	FOCUS 2004
Hedges, windbreak	late	evaluation of literature	75 %	FOCUS 2004

(* based on Schulz et al 2007)

Based on the inhomogeneous experimental data available the *FOCUS group on landscape and mitigation* suggested reduction factors of 25 % to 75 % dependent on the leaf stage (see bottom of Table 4-11).

Based on this information a linear model was developed for the geo-referenced probabilistic model which gives a defined reduction factor for each Julian day based on the minimum and maximum values suggested by FOCUS (2007). The respected dates when the model reaches its minimum and maximum reduction are shown in Table 4-12. A graphical representation of the model is presented in Figure 4-13.

Table 4-12: Benchmark figures of the linear model describing daily reduction factors

Model endpoint	Date	Phenology
Emergence of leaves	1st April	cherry blossom
Maximum coverage	15th June	midsummer
Beginning of leaf fall	1st October	beginning of autumn
Leaf fall completed	15th November	winter

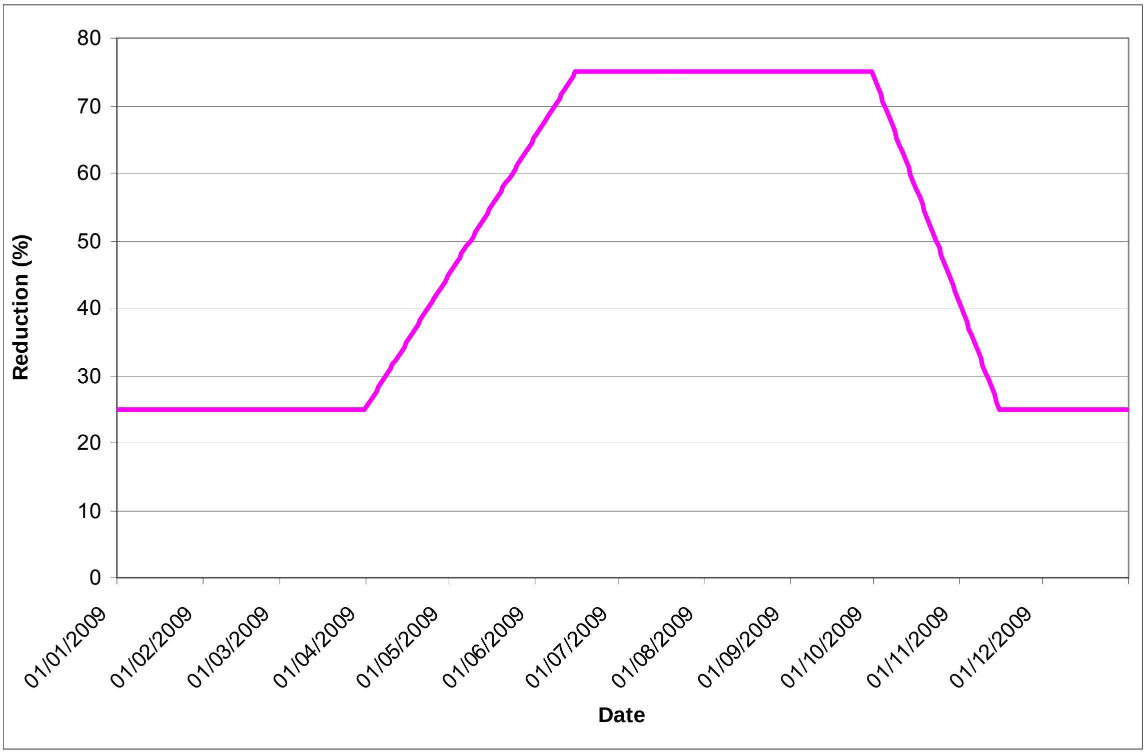

Figure 4-13: Simulation of reduction factors dependent on the season of the year

Of course, the dates presented in the model may vary between regions and from year to year, but these dependencies are not considered.

4.2.5.3 Sensitivity of the drift reduction factors due to waterside vegetation

The drift loadings into surface water could be reduced up to a factor of 4 if shielding vegetation was considered in the geo-referenced system. This process would therefore be a key factor when drift reduction is discussed at the landscape scale.

The model as it has been defined above does not have a probabilistic component. It is not expected that introducing a certain variability of the reduction rates will significantly improve the calculation. Of course, the concrete dates when pesticides are applied may change from year to year depending on the weather conditions but the application dates are mainly driven by the phenology (e.g. blossom) rather than fixed dates. It can be assumed that a certain crop development for the target crop is somehow correlated with the development of the shielding vegetation at the same location.

If, nevertheless a year to year variability of one week is assumed for the application date the effect on the calculated initial concentration in surface water is rather small compared to other uncertainties (see Figure 4-14). Considering the additional effort that is necessary when such an uncertainty is introduced (because of the additional number of simulations that have to be performed) it is recommended to simply use the mean values of the models to consider the shielding effect of waterside vegetation.

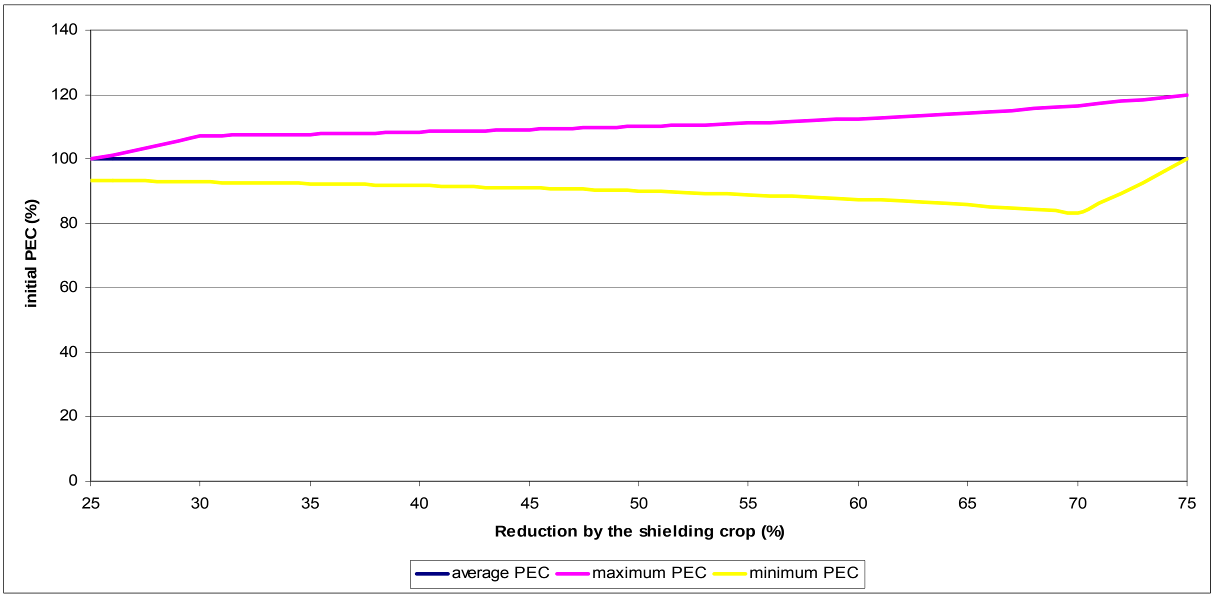

Figure 4-14: Variability of the PEC when considering an uncertainty of 1 week for the application date

To what extent waterside vegetation actually reduces the drift loadings also depends on the occurrence of this type of vegetation in the agricultural area and the representation of these landscape elements in available data sets. This issue is also discussed in chapter 5 when exploring uncertainties of underlying spatial data.

The ATKIS-Perspective

A hotspot-analysis was carried out for all water bodies near hops in Germany. In total, 650 km of rivers and ditches were considered; the segment length was 25 m.

A PEC_{ini} distribution per water body segment and wind direction was computed for using the following formula:

$$PEC_{ini\ i\,WD\,j} = \frac{AR \cdot drift}{Vol_i} * RED \qquad (1)$$

$PEC_{ini\,i\,WDj}$	PEC of the water segment *i* at exposure from wind direction WR *j*	[µg/l]
AR	application rate	[kg/ha]
drift	drift deposition rate, relaed wind direction WD_j	[-]

Vol_i	volume of the water body from segment	[l]
RED	*r* eduction factors, e.g. for drift reducing vegetation	[-]

The calculation of the spray drift deposition rate has been considered according to chapter 4.2.4.

We analysed all water body segments for $PECp_{90}$ > RAC of all eight wind directions. The results of the analysis are given in the following table.

Table 4-13: Hotspot-length dependent on shielding vegetation calculated for a 10 m buffer regulation.

Reduction factor [%]	Without shielding vegetation [km]	With shielding vegetation [km]	Difference [%]	Difference [km]
25	133	133	0.1	0.1
50	133	132	0.7	1
90	133	131	1.5	2

It has to be noted that looking at the $PECp_{90}$ a of water body segment a hedgerow has to be present at both sides of the water body to reduce the $PECp_{90}$. This is not necessarily the case when a lower percentile or statistical value (PEC_{mean}) is considered.

The HR-Perspective

In order to clarify the question to what extent the consideration of shielding crops will reduce the drift loadings and the calculated hotspots we included geo-data derived from the GeoPERA-project. Similar as in chapter 4.2.2 high resolution geo-data coming from the hop growing region “Hallertau” (see Figure 4-5) were selected for this analysis. Water body segments with 10 m length were analysed by using high resolution geo-data as ortho images and administrative topographical data from the state of Bavaria (“Feldstücksdaten"). The combination of hr-geodata and the Atkis-DLM2 dataset yielded in total 564 km of streams within a 150 m-buffer around hops areas. 300 km of the net were included in the analysis using hr-geodata (60 %). Some information about the structure of the test area is given in Table 4-14. A detailed documentation of the methodology can be found in the reports sent to the UBA (Trapp et al. (2008) and Wagner et al. (2007)). The use of 10 m segments in the hr-analysis based on the former GeoPERA-project, where it was decided to use 10 m segments when using hr-geodata and 25 m segments when using the ATKIS-data taken into account the different geometrical resolutions of these two datasets.

All following information are derived by analysing the 300 km water segments based on a high resolution landscape classification.

As is shown in Table 4-14, the 10 m buffer area around the 300 km water body segments covers an area of about 550 ha, including more than 100 m three dimensional biotopes (nearly 20 %) and nearly 27 ha of hop fields.

Table 4-14: Characterization of the example area “Hallertau” (only hr-region, see Figure 4-5)

	10 m-buffer [ha]	Buffer area [%]
Buffer area	550.45	100.00
Biotopes	109.53	19.90
Hops	26.95	4.90

The 90th percentile of the spatial PEC distribution was used considering equal distribution of the eight wind directions according to the methodolgy used in the GeoPERA-project. Despite that the GeoRisk approach is based on the presence of hotspots as decision criterion instead of a 90th percentile from a spatial distribution (see chapter 9), the use of the percentile here is considered reliable because it is used only to analyse the principal effect of waterside vegetation.

The hotspots (RMS) were calculated based on a 1000 m moving window (hotspot-criteria: PEC exceeding RAC in 10 % of segments or once by 10-times).

The results for a buffer of 10 m around the water bodies are presented in the following tables and figures.

Table 4-15: PEC_{mean}: Results with a 10 m-buffer around water bodies

Reduction factor [%]	Without shielding vegetation [µg/l]	With shielding vegetation [µg/l]	Reduction effect due to shielding vegetation [%]
25	3.99	3.05	23.56
50	3.99	2.28	42.86
75	3.99	1.77	55.64

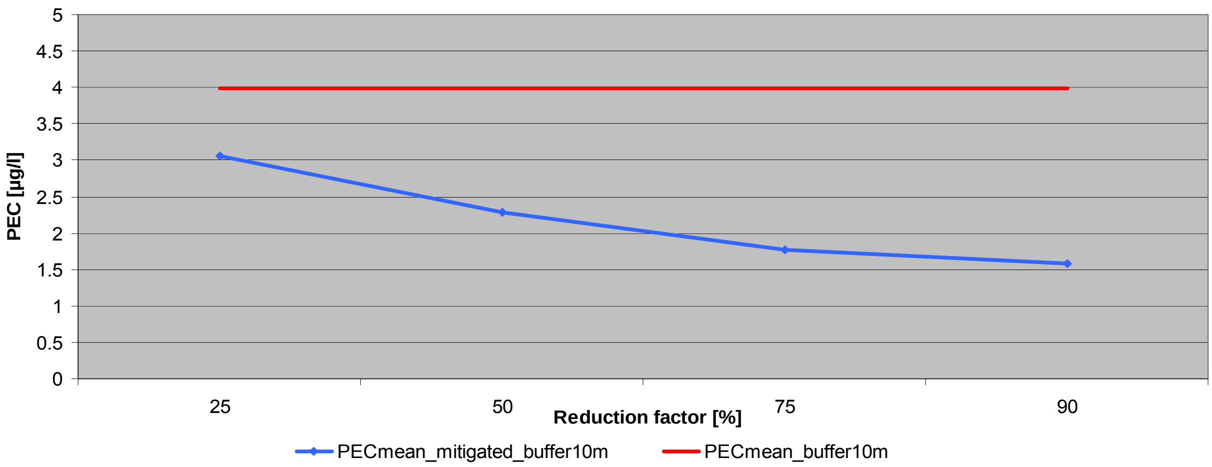

Figure 4-15: Reduction of PEC_{mean} in the Hallertau dataset dependent on the actual shielding factor based on a 10 m spraying buffer zone (red line: not shielding)

As shown in Figure 4-15 shielding vegetation can reduce the mean predicted environmental concentrations in the Hallatau area to about 55 % if the maximum reduction factor of 75 % is assumed.

Table 4-16 gives some information about the consequences for the number of segments with concentrations above the RAC. If a maximum reduction of 75 % is assumed a reduction of segments above the RAC from 11.48 km to 9.43 km is calculated.

Table 4-16: Segments with PEC_{mean} > RAC (10 m buffer)

Reduction factor [%]	Without shielding vegetation [km]	With shielding vegetation [km]	Difference [%]	Difference [km]
25	11.48	10.04	12.54	1.44
50	11.48	9.6	16.38	1.88
75	11.48	9.43	17.86	2.05

4.2.5.4 Conclusions: water side vegetation

Besides the geo-referenced distance between water body and field the existence of shielding crops shows the greatest impact on the local PEC values. However, as experimental data show there is significant variation of existing drift reduction factors dependent on

- the local situation, as landscape structure varies between landscapes.
- spatial and thematic resolution of the underlying landscape models (e.g. ATKIS BDLM vs. hr data), due to limits of detection, classification and nomenclature.
- the choice of the PEC percentile considered at segment level (e.g. PEC_{mean}, $PECp_{90}$). When analysing the effect of drift reduction due to shielding crops referring the 90th percentile then the criteria PEC>RAC is hardly influenced whether there is a hedgerow present at both sides or just on one side of the water body.

It is therefore suggested to consider reduction rates in the geo-referenced software based on the recommendation of FOCUS 2004 using a simple linear model.

4.2.6 Parameter emerse vegetation and shielding herbs

A reduction of the spray drift loading due to emerse vegetation in the water and shielding herbs above the water is not considered in the current (worst case) risk assessment, although it is expected that this has a significant effect on the drift entry at least in the warm season.

4.2.6.1 Calculation of drift entries considering the emerse vegetation

Drift entries dependent on shielding vegetation could be calculated rather simply using the same approach as explained in the previous section for the water side vegetation. The equation could be also used in geo-referenced systems without additional modifications.

$D_{red} = D_0 * d_{red}$

d_{red}: drift reduction caused by emerse vegetation or shielding herbs above the water (-)
D_0: standard drift percentile with classic technology (%)
D_{red}: reduced drift percentile if waterside vegetation is present (%)

The consideration of the waterside vegetation will only affect the initial loading but not the time dependency of concentrations in surface water.

4.2.6.2 Availability of input parameters

A prerequisite for considering waterside vegetation in the geo-referenced model is the availability of the following two factors. First, information on the presence of waterside vegetation in the system and secondly, drift reduction factors as a function of vegetation type and/or the leaf stage.

The structure of both target crop and plants in the margin between the application site and the surface water can have a large influence on rates of deposition to surface waters.

Van de Zande et al. (2000) assessed spray drift when spraying a sugar beet crop. The field margin neighbouring the cropwas planted with a 1.25 m wide strip of *Miscanthus* (Elephant grass) with heights varying between not planted (0 m), at crop height (0.5 m), 0.5 m above crop height (being sprayer boom height, 1.0 m) and 1 m above crop height (1.5 m). Application was performed with a conventional and an air-assisted sprayer. The height of the vegetation had a clear effect on spray drift deposit. At 3-4 m distance from the last nozzle spray deposit decreased significantly with increasing heights of *Miscanthus*. Cutting *Miscanthus* to the same height as the sugar beet resulted in 50 % spray drift reduction compared to spray drift occurring at the same distance but without shielding vegetation. Spray drift was reduced by 80 and 90 % with *Miscanthus* 0.5 and 1.0 m above crop height, respectively.

In other experiments performed by de Snoo and de Wit (1998) was shown that a 3 m buffer zone covered by vegetation resulted in a decrease of the drift deposition by a minimum of 95 %. Wind tunnel experiments done by Miller and Lane (1999) showed drift reduction of 34.7 % caused by a grass and wild flower mixture compared to short grass of 20 cm height.

The mitigation afforded by a margin comprised of grass and wild flower mixture with a base canopy height of 0.7 m with elements extending to 1.3 m high was of the order of 60 – 85 % relative to drift observed with a 0.15 m mowed grass margin (FOCUS 2004).

An inventory of suggested reduction factors up to 67 % for the emerse vegetation and up to 90 % for small bank vegetation has been generated by Schulz et al. (2007). Reduction factors were mainly dependent on the leaf stage (25 % vegetation: 27.1 % reduction, 80 % vegetation: 67.2 % reduction).

4.2.6.3 Sensitivity of the drift reduction factors

As shown in Table 4-11 the drift loadings into surface water could be reduced up to one order of magnitude if emerse vegetation is considered in the geo-referenced system. Similar as the waterside vegetation this process could therefore be a key factor when drift reduction is discussed on the landscape scale.

However, in contrast to the waterside vegetation the pesticides could nevertheless reach surface water after rainfall events because the shield is located directly above the surface water. Dependent on pesticide specific wash-off factors and on additional individual disappearance processes (photo degradation and volatilisation from plant surfaces, plant uptake) smaller or larger amounts could still reach the surface water at a later stage

4.2.6.4 Conclusions: emerse vegetation and shielding crops

As many surface waters are shielded by vegetation during the vegetation period it is important to consider the respective drift reduction. There is sufficient data available how efficiently these shielding process works. However, even if the existence of this shield can be demonstrated by geo-referenced data it may be difficult to consider this process for the generic examination. The reason is that the process is also substance specific because additional disappearance processes have to be considered to ensure that the compound will not reach surface water caused by the first rainfall event after spraying.

It is therefore recommended to consider the effect of shielding vegetation above the surface water (emerse vegetation and herbs) on a substance basis similar as hydrolysis or photolysis though it is originally a discharge route.

That could be done similar as current consideration of runoff in the model EXPOSIT (Winkler 2005) by defining a constant time between spraying and the wash-off event.

However, the analysis of existing literature demonstrated that additional research in this field is needed.

A discussion on the shifiting or neglecting the mowing of the embankment can be found in chapter 9.4.

4.3 Calculation of entries via volatilisation

In contrast to spray drift possible entries via volatilisation have to be considered dependent on pesticide properties such as vapour pressure or Henry’s constant. Presently in the German registration of pesticide these burdens are estimated using the software tool EVA 2.0 [Koch 2005]. In the following it is analysed whether in the geo-referenced assessment scheme entries caused by volatilisation during and shortly after application could be considered similarly.

4.3.1 Parameter vapour pressure

The vapour pressure is a substance and temperature specific gas pressure, at which the substance’s vapour phase is in equilibrium with its liquid or solid phase. Descriptively explained, the vapour pressure is the atmospheric pressure at a given temperature at which liquids start boiling.

The assessment tool EVA 2.0 uses the vapour pressure as key parameter to describe the extent of volatilisation.

4.3.1.1 Calculation of the entries caused by volatilisation

The calculation of the surface water entries caused by volatilisation can be calculated based on the method implemented in EVA. All pesticides are ranked in four classes with regards to their vapour pressures. For each class a “volatilisation percentile” has been defined (similarly to a drift percentile), which is related to total deposition after 24 h at a distance of 1 metre from the target area (see Table 4-17).

Table 4-17: Deposition by volatilisation according to EVA 2.0 at a distance of 1 m [Koch 2005]

Vapour pressure vp	Meaning	Soil application % of application rate	Plant application (field crop) % of application rate	Plant application (orchard/vine/hop) % of application rate
vp < 10^{-5} Pa	non volatile	0	0	0
10^{-4} Pa > vp >= 10^{-5} Pa	semi volatile	0	0.267	0.534
5 * 10^{-3} Pa > vp >= 10^{-4} Pa	semi volatile	0.221	0.663	1.326
vp >= 5 * 10^{-3} Pa	volatile	1.555	4.665	9.33

The load into the surface water caused by volatilisation can be calculated according to the following equation:

V_1 = Ar * [Ic * Va (Vp, crop) + (100-Ic) * Va(Vp, soil)] / (100 * 100)
V_1: Deposition by caused volatilisation at 1 m distance to the target area (g/ha)
Ar: application rate [g/ha]
Ic: crop interception (%)
Va: crop and substance specific volatilisation fraction at 1 m distance to the target area (%)

The respective crop interception is given by the BBCH-stadium and is summarised in tables (for example in Exposit (Winkler 2005)).

The distance dependency of V_1 is calculated according to following equation:

V(d) = V_1 * exp[-0.05446 * (d-1)]
V(d): deposition at distance d to the target area (g/ha)
V_1: deposition at distance 1 m to the target area (g/ha)
d: distance to the target area (m)

In contrast to the presented algorithm EVA is additionally considering a certain reduction of the application rate caused by drift losses. It is based on a fixed field size of 1 ha.

As this correction is only in essentially changing the original dose it is recommended to neglect this term.

Similar to the calculation of the drift entry this process would primarily influence the initial concentration in surface water bodies. However, the vapour pressure would also influence the time dependent concentrations in surface water because dependent on wind speed and stream velocity substances may leave the water body into the air.

4.3.1.2 Availability of vapour pressure

The parameter vapour pressure is generally available for all pesticides.

4.3.1.3 Sensitivity of vapour pressure

Compared to volatilisation spray drift will normally dominate the entries into surface water bodies. However, if a volatile compound is applied directly to the crop (interception: 100 %) the situation could be vice versa as demonstrated in Figure 4-16. The calculated deposition caused by volatilisation is significantly higher than spray drift in vineyards. Even compared to orchard higher amounts are deposited at least close to the target area.

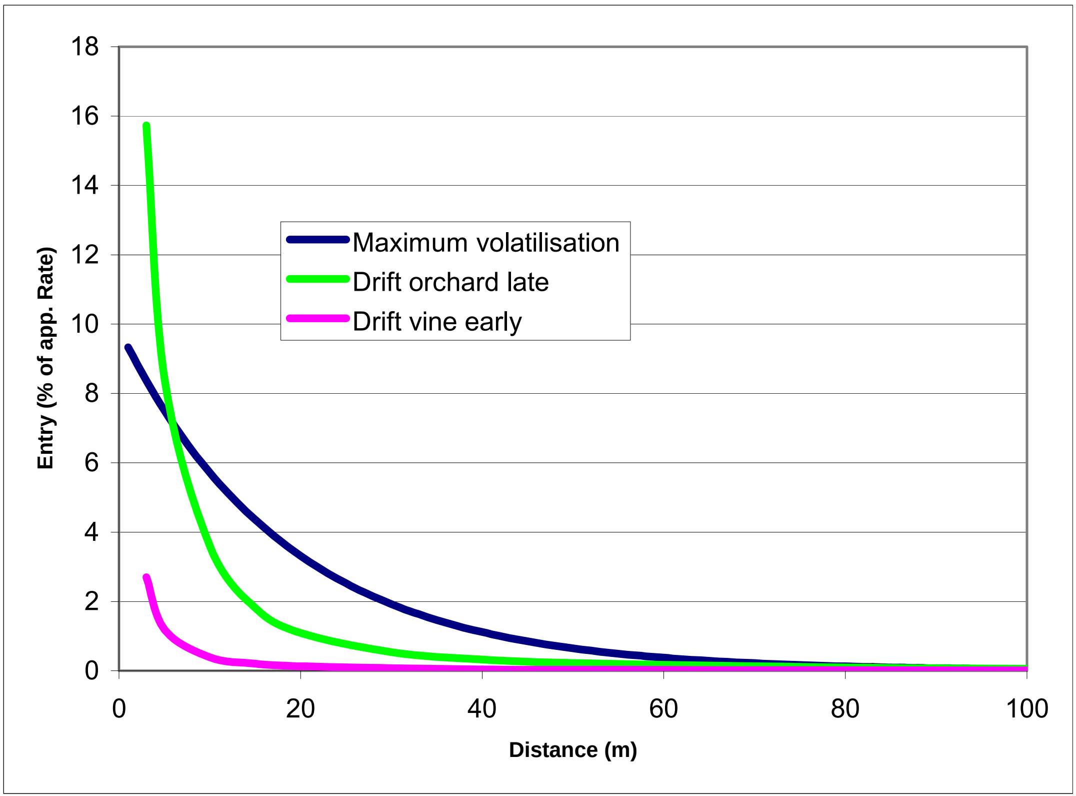

Figure 4-16: Comparison of entries caused by spray drift and volatilisation for a volatile substance

4.3.1.4 Conclusions: volatilisation entries

For the generic examination volatilisation cannot be considered as this process is highly substance specific.

However, according to the model EVA deposition caused by volatilisation can be significantly higher than respected deposition caused by spray drift at least for volatile compounds.

It is therefore recommended to consider this entry route on a substance base only.

The process can be implemented rather easily as an additional load such as spray drift by selecting the crop- and substance specific deposition percentile together with the BBCH-stage at the time of application. Based on this information the deposition at 1 m can be calculated as a constant rate. Dependent on the actual distance to the surface water body this amount has to be corrected within the geo-referenced system using the decline function of EVA.

4.4 Calculation of surface water concentrations in static water bodies

4.4.1 Parameter water depth

The water depth is a key parameter for the calculation of surface water concentration caused by spray drift. Traditionally, the depth of the water body is considered by a single number of 30 cm. Though the parameter is not directly available in spatial databases it can be estimated based on the width of surface water bodies.

The calculation of surface water concentrations dependent on the water depth is rather simple as shown in the following equation for initial concentrations:

$$C_0 = \frac{App \cdot D}{depth \cdot 100}$$

C_0: initial concentration in surface water (µg/L)

D: drift deposition rate as percent of the application rate (%)

App application rate (mg/m²)

depth: depth of the surface water (m)

4.4.1.1 Availability of the input parameters

The water depth is not directly a geo-referenced parameter. In the German as well as in the European pesticide registration process a relationship between width (w) and depth (d) of 3.3:1 (1 m width and 0.3 m depth, FOCUS standard ditch) for lentic surface water bodies is assumed.

Only a very small set of measurements of the w/d ratio for ditches are available in literature. Three studies from Germany and The Netherlands present data for ditches (Schäfers et al. 2006, Nijboer et al. 2003, Golla et al. 2007 in Table 4-18), which gives medians of the width/depth ratio of 6.2, 8.3, and 5.6, respectively. These results suggest that real world ditches have less water volume compared to the FOCUS standard ditch scenario. This may lead to an overestimation of the depth and thus, an underestimation of the PECini. Therefore more data on the w/d ratio for lotic water bodies is necessary in order to gain a regionalized view on this important parameter to be used in the geo-referenced PRA risk assessment of lentic water bodies. However, considering the very small database on measured width/depth ratios for real water bodies in Germany a mean ratio of 6.6 : 1 for ditches seems to be a passable starting value at the moment based on the ratios obtained in the Altes Land, Brandenburg and The Netherlands (Table 4-18).

According to the literature review for streams in Germany (and furthermore some results from France), the w/d ratio seems to be higher and approximately around 10 (see Table 4-18). Nevertheless these values are only first approximations and have to be refined and validated, respectively by further measurements in different landscapes in Germany. For the new development of exposure models for streaming water system these differences have to be respected in the context of a geo-referenced PRA (see chapter 8).

Table 4-18: Width/depth ratios for different water bodies (ditches considered as lentic; streams/rivers considered as lotic))

Data set	n	Median ratio w/d	90th percentile
FOCUS standard ditch	1	3.3	-
Altes Land ditches (Schäfers et al. 2006)	40	6.2	15.2
Dutch ditches (from Brock et al. in press, after Nijboer et al. 2003)	?	8.3	16.7 (?)
Streams around Braunschweig (Pantel 2003)	40	8.7	26.5
Streams around Braunschweig (from Wogram in press, after Wogram 1996)	15	9.8	< 32.5
Streams in Südpfalz (Schulz et al. 2007)	39	9.0	23.0
Streams in Hallertau (Claßen, pers. comm.)	14	7.2	12.7
Ditches in Brandenburg (Oderbruch) (Golla et al. 2007) Water body width (mean)= 2.9m, Width of ditch bank (mean) = 1.9m	39	5.6	7.6
Streams in Bodensee region (Golla et al. 2009)	42	13.29	30.00
Streams in SN orchard regions (Golla et al. 2009)	13	9.65	14.00
Streams in MV orchard region (Golla et al. 2009)	8	13.67	27.00
Streams in BB orchard region (Golla et al. 2009)	13	11.70	33.33
Streams in TH/ST orchard regions (Golla et al. 2009)	31	13.69	22.00
Stream reaches in France (Loire, Rhone, Garonne) mean Q=3,31 (m³/s), B/T for Q50, (Lamouroux, Capra, 2002)	34	31.2	Max. 51
Stream Kühbach (Bayern), Station Danzersäge, B/T=f(Q) from n=7 discharges; Q= 1 - 5 m²/s, (Lamouroux et al. 1992)	1	11.1	B/T = 10,7*Q0,05
16 Rivers in France; Q50(actual) = 0,90 m²/s, (Lamouroux et al. 1995)	16	26.1	Max. 39

4.4.1.2 Conclusions: water depth

The water depth is a key parameter when estimating surface water concentrations caused by spray drift. For the hotspot identification of static water bodies, where no water exchange (i.e. dilution of pesticide concentration) is assumed, a width/depth ratio of 6.6 : 1 should be applied in the geo-referenced assessment. This value is approximately the average of the mean width/depth ratios for ditches in Table 4-18 (data from Schäfers et al. 2006, Nijboer et al. 2003, Golla et al. 2007), the value finally is fixed to 6.6 because this is the doubled value of the FO-CUS scenario.

With respect to the highly divers values on measured width/depth ratios (Table 4-18) for rivers and streams, the authors are not in the position to propose a scientifically based single value for the width/depth ratios for ditches and streams. In case of the implementation of a geo-referenced PRA further measurements in different landscapes combined with an GIS-based extrapolation will be necessary to determine distinct width/depth ratios for rivers. This GIS analysis has to estimate not only the variables width and depth but also the flow velocity and the discharge volume..

4.4.2 Parameter water body profile

According to Bach (2004) natural brooks and river in the flatland and low mountain regions in Germany typically has an U-shaped cross section (valid for low flow and average discharge conditions). For reasons of (i) mathematical simplification of the calculations for the dynamic exposure model (cf. chapter 4.6 and 8), and (ii) conceptual consistency with the "static ditch" approach, for the new geo-referenced probabilistic model it is recommended to use the simple box geometry (rectangle) profile as assumed water body cross section.

To judge the effect of different cross sections on the target variables of the exposure modeling, for three idealized standard forms of a cross section (rectangular, trapezoid, and segment of a circle) the results for PEC_{ini} and concentration maximum (c_{max}, Figure 4-17) are calculated with otherwise identical values for spray drift deposition, flow velocity and area of the cross section (i.e. identical discharge).

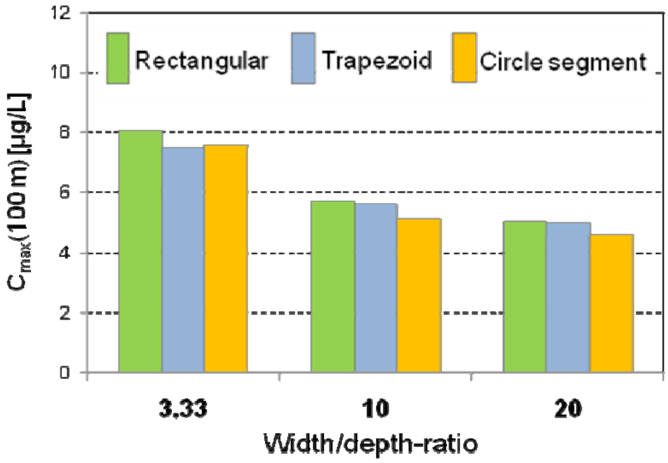

Figure 4-17: Concentration peak in rivers with identical cross section (0.1 m²) and flow velocity (0.2 m/s) but three types of cross section and thus different width/depth-ratios. C_{max} is the concentration peak for a river segment 100 m downstream the segment of deposition, calculated for a drift deposition of 10 mg/m² water surface with respect to the hydrodynamic dispersion of the initial PEC_{ini}.

As expected, PEC_{ini} is highest (results are not presented in a diagram) for a river with a section of a circle segment, the most "flattened" cross section (largest surface area per average depth for a given width/depth-ratio). But the dynamic approach demonstrates that solely the initial concentration is not sufficient to capture the effect of section type. This effect is overlaid by a much more intensive hydrodynamic dispersion in a river with trapezoid or even circle segment cross section (for details of calculation rf. chapter 4.6.). Looking at c_{max} the effect of cross section is turned around: The rectangular profile, the one with the lowest PEC_{ini}, has the highest c_{max} after 100 meter of streaming (Figure 4-17). The ranking of cross section standard profiles "rectangular < trapezoid < segment of a circle" with respect to dilution of the initial concentration in streaming waters is irrespective of the flow velocity, the discharge volume, and the width/depth-ratio. Hence, the use of a *rectangular* cross section as standard profile type is the most conservative (protective) assumption in the context of dynamic exposure modeling.

Furthermore, the differences in the column heights in Figure 4-17 are small. Thus one can conclude that the variance and uncertainty of results caused by a maybe wrong assumption "what is the correct river profile" are of minor relevance compared to other factors (e.g. distance of treated field, discharge volume etc.).

4.5 Volatilisation from surface water

Substances, which may on one hand enter surface water during and shortly after application via volatilisation because of their physico-chemical properties (e.g. vapour pressure), on the other hand also tend to volatilise out of the surface water.

Apart from the physico-chemical properties of the substance this process is influenced mainly by wind speed and the water flow.

This process is currently not considered in the registration procedure.

The volatilisation rate (respectively the volatilisation half life) in water can be calculated based on the total mass transfer coefficient K_L based on following equation.

$k_v = K_G / t$ $\qquad$ $DT_{50v} = \ln(2) / k_v$

k_v: volatilisation rate from surface water (1/d)
DT_{50}: volatilisation half life in surface water (d)
K_G: total gas phase mass transfer coefficient from surface water (m/d)
t: time (d)

The total mass transfer coefficient K_L is calculated based on a substance specific term (Henry's constant) and a scenario specific term (flow velocity, wind speed) according to following equation:

$1/K_G = 1/k_g + H'/k_l$
$H' = H / R\,T$

K_G: total gas phase mass transfer coefficient from surface water (m/d)
k_g: gas-phase exchange coefficient (m/d)
H': nondimensional Henry's law constant (-)
k_l: liquid-phase exchange coefficient (m/d)
T: Temperature (K)
H: Henry's constant (J/mol
R: gas constant, J/(K mol)

Southworth developed a method to calculate both the gas phase and the liquid phase exchange coefficient based on aromatic hydrocarbons [Southworth 1979). But the following equations can be used also for other chemicals:

k_g = $1137.5\,(u + v)\,\sqrt{18/M}$

k_g: gas-phase exchange coefficient (m/d)
u: wind speed (m/s)
v: flow velocity (m/s)
M: molecular mass (g/mol)

k_l = $23.51\frac{v^{0.969}}{z^{0.673}}\sqrt{32/M}$ (if u< 1.9 m/s)

k_l = $23.51\frac{v^{0.969}}{z^{0.673}}\sqrt{32/M}\cdot\exp[0.526\cdot(u-1.9)]$ (if 1.9 m/s < u < 5 m/s)

k_g: liquid-phase exchange coefficient (m/d)
u: wind speed (m/s)
v: flow velocity (m/s)
M: molecular mass (g/mol)
z: depth of the surface water (m)

4.5.1 Availability of necessary input parameters

The substance specific and temperature dependent parameter Henry's law constant (or alternatively vapour pressure and water solubility) is generally available for all pesticides, at least for standard temperatures. However, for the calculation of volatilisation from water bodies three more scenario parameters (depth of the surface water, wind and flow velocity) must be known. The depth of the surface water is also generally available and it is used for calculation of initial concentrations. So far the flow velocity is not used in the current system, as the worst case scenario is based on a static water body. Unfortunately, a number unequal zero is needed in the calculation. It is suggested to use a low flow velocity of 0.01 m/s for static water bodies. Of course, for the calculation of streams in a future system explicit flow velocities must be known, but this parameter can also be made available (see chapter on modelling streams).

For the wind speed it is suggested to use averaged regional values of local weather stations. More precise information would be not reasonable for these calculations.

The volatilisation may also be influenced by the presence of emerse vegetation though this influence is not directly mentioned in the equation. However, information on vegetation could be considered indirectly by additional estimation of its influence on the water temperature stream flow velocity and local wind speed.

4.5.2 Sensitivity of the reduction by volatilisation

Volatilisation will generally not influence the initial concentration in the surface water. Only time dependent concentrations can be reduced by this process. An impression about the expected reduction rates caused by volatilisation for static water bodies is presented in Table 4-19. For volatile compounds and standard conditions (water depth: 30 cm and wind speed 3 m/s) half lives are calculated in the range of days. For shallow surface waters (e.g. 10 cm) the residence time can be significantly smaller (few hours) for the same substances.

In contrast to static water bodies, considerably shorter volatilisation half lives are calculated for streams.

Table 4-19: Calculated half lives in static water bodies caused by volatilisation [Southworth 1979]

Wind speed (m/s)	Water depth (cm)	Vapour pressure (Pa)	Water solubility (mg/L)	$DT_{50volat}$
3	30	0.01	1	3.2 d
3	30	0.001	1	11.4 d
3	30	0.01	10	11.4 d
3	30	0.001	10	94 d
1	10	1	1	15 h
3	10	1	1	9 h
1	30	1	1	4 d
3	30	1	1	2.2 d

* flow velocity 0.01 m/s

As long as in the geo-referenced probabilistic risk assessment model only static water bodies are considered it is not meaningful to consider volatilisation out of the surface water as an additional loss process. However, when also streams are simulated volatilisation could be a significant loss process, at least for the volatile compounds. Local volatilisation rates can be calculated in the probabilistic GIS model based on the Henry's law constant and three spatial parameters, the depth of the surface water, the local windspeed, and the flow velocity according to the above described equations. The availability of spatial data on water depth and flow velocity and some approaches to overcome this problem are discussed in chapter 4.6.3 and chapter 8. For the wind speed an average value (or a respective worst case value e.g. 10^{th} percentile) from a local weather station would be sufficient because in contrast to the very fast process spray drift during application the time scale for this disappearance process are hours rather than seconds.

The volatilisation rate could then be added to the other loss process considered in the model (e.g. the biodegradation rate).

4.5.3 Conclusions: volatilisation out of surface water bodies

For the generic examination volatilisation cannot be considered as this process is highly substance specific.

However, according to the model exemplary calculation presented for volatile compounds discharge caused by volatilisation can be significantly higher than microbial degradation.

It is therefore recommended to consider this discharge route on a substance base similar as hydrolysis or photolysis.

It would be principally possible to consider this process at least on a scenario based method based on constant wind speed.

It is however not possible to consider this process outside the GIS-software (e.g. by an overall reduction rate added to microbial degradation), because volatilisation is extremely dependent on water depth.

4.6 Model concept for the geo-referenced probabilistic assessment for streaming waters

4.6.1 Starting point

According to field observations (cf. the analyses from Golla et al. 2009; Trapp et al. 2009), the morphology of surface waters – i.e. natural rivers in regions with permanent crops – is typically characterized by the following two features:

Normally surface water *flows*. Stagnant bodies of water (with the exception of lakes) are rare in undulating landscapes.

The width/depth ratio: according to field surveys of water bodies, the average B/T ratio for small rivers not wider than 2 m is about 10 or larger.

It can therefore be concluded that accepting the current standard registration scenario for a width/depth ratio of 3.33 (i.e. a dilution of 300 L/m²) is not conservative regarding the resultant predicted environmental concentration (PEC), because 3.33 corresponds to only about the 10^{th} percentile of B/T ratios in actual rivers (for more data refer to Table 4-14, Chapter 4.4.1.3)

When considering a stagnant water body, the actual specifications of most surface waters, including the internal transport and transformation processes, become unreliably simplified. If exposures were evaluated or measured instead on a geo-referenced basis, then this would by necessity demand the highest level of convergence with the actual morphology of rivers, which is usually affected by channel hydrology in most cases.

Based on these conclusions, the exposure model structure for the geo-referenced probabilistic assessment of PEC spray drift in streaming waters can therefore be outlined.

4.6.2 Fundamentals of exposure modelling for streaming water

A model for assessing exposure in flowing water must take into consideration the following core system characteristics:

Discharge in a river network system is spatially and temporally variable.

Pesticides are applied sequentially to the fields alongside a stretch of river, therefore resulting in varying frequencies (0, 1, …, k) of spray drift deposition on the individual portions of water.

Due to hydrodynamic dispersion concentration peaks in water with increasing distance from the pesticide's point of entry become more dispersed and simultaneously level out (see chapter 2.3).

Spray drift deposition and the ecotoxicological effect are spatially and temporally decoupled: Thus, pesticide deposition on a single part of the water stretch can have ecotoxicological effects on a large number of river segments downstream the affected part.

4.6.3 Fundamentals of river hydrology

In order to model the water quality in rivers an understanding of the fundamentals of hydrology is required. According to Chapra (1997) knowledge of the following parameters is necessary for quality modeling: discharge, flow velocity, dispersion, flow depth, channel width, and river bed gradient (slope of energy line). This enables to describing the transport and transformation processes within the body of water. Therefore a hydrodynamic model is an essential part of a river quality model.

The starting point for connecting these parameters is the flow velocity according to the **Gaukler Manning Strickler** equation.

$$v_{fl} = Q / A = k_{St}\, R^{2/3}\, J^{1/2} \quad [m/s] \qquad (1)$$

whereas:

v_{fl}	mean flow velocity [m/s]
Q	discharge (flow volume) [m³/s]
A	cross-sectional area of flow [m²] = channel width *b* * flow depth *h*
k_{St}	roughness coefficient [$m^{1/3}$/s]
R	hydraulic radius [m]
J	river bed height gradient [m/m]

In order to generate geo-referenced variables for the necessary hydraulic parameters, i.e. estimated values for each river segment, the following procedure is recommended.

(i) River bed gradient J

This can be calculated as a mean slope for longer river sections by overlaying a digital elevation model (DEM) onto a digital map of the surface water network. The stretch for which a river gradient is averaged should have a minimum length of approx. 1 km, over which J can be calculated. Additionally, check the results in order to place the parameters for J in low mountain ranges (outside the floodplains of larger streams) as follows: $0.001 \le J \le 0.05$.

(ii) Discharge Q

Basic assumption: the flow volume, Q_x, at an arbitrarily chosen point, x, of a river stretch is proportional to the river's watershed, $FN_{x,}$ up to this point.

$$Q_x \sim FN_x \qquad (2)$$

If it is further assumed that each surface unit of the river watershed with the same area-specific runoff rate *q(t)* at time *t* adds to the runoff, then the flow volume at a random point of the river at a given time can be described as:

$$Q(t)_i = q(t) * FN_i \qquad (3)$$

whereas:

$Q(t)_i$	discharge at time *t* in segment *i* [m³/s]
q(t)	area-specific runoff rate in the watershed at time *t* [$m^3\, s^{-1}\, km^{-2}$]
FN_i	watershed area up to segment *i* [km²]

<u>Wanted:</u> FN_i and q(t), which can be calculated as follows:

a) Watershed area FN_i. Assumption: The length of the river section in a landscape is proportional to the watershed area from which it drains. During the initial approximation, it can be assumed that the river network density (RND) within a larger watershed is nearly constant:

$$RND = \text{River segment} / \text{Watershed area} \; [km/km^2]$$

$$RND = \Sigma\, Segm_{Wts} / FN_{Wts} = \Sigma\, Segm_i / FN_i \qquad (4)$$

According to formula (4) FN_i can be calculated from:

$$FN_i = \Sigma\, Segm_i / RND = \Sigma\, Segm_i * \Sigma\, FN_{Wts} / \Sigma\, Segm_{Wts} \quad [km^2] \qquad (5)$$

Prerequisites: map(s) with:

- borders around the river’s (partial) watersheds FN_{Wts}
- the length of all river sections (number of segments) $\Sigma\, Segm_{Wts}$ in the watershed FN_{Wts}
- the length of the river section (number of segments) $\Sigma\, Segm_i$ from upstream down to segment *i*
- Important prerequisite: topologically correct river network with as few gaps as possible

b) Area-specific runoff rate *q(t)*: regionally specific values for a watershed FN_{Wts} can be calculated by analyzing runoff measurements (water gauge measurements) for river basins.

For smaller watersheds with minimal differences in elevation, relatively evenly dispersed precipitation as well as relatively uniform ground compositions and aquifer characteristics can be assumed during the initial approximation; consequently, the area-specific runoff rate *q(t)* which is calculated at the water gauge as an average value for the entire watershed can truly be expressed as a useful estimation for the runoff rate of partial watershed areas.

Knowing the area-specific runoff rate *q(t)* and the size of the watershed FN_i, enables to estimating the drainage *Q(t)* for each segment *i* according to formula (3).

(iii) Roughness coefficient k

Initial approach: a mean value of k = 15 should be set (cf. e.g. Schröder, 1979).

If more empirical values from field measurements are available, a possible relationship between *k* and *J* or rather *k* and *b* can be tested.

(iv) Channel width b and flow depth h

A river’s width and depth can be calculated from equation (1) if Q, J, and k are known. They can be introduced as the following assumptions:

Width/depth relationship = 10 (median of the previously calculated field measurements),

Rectangular (box-shaped) cross-sectional flow.

Equation (1) restated is:

$$B = 4.414 * Q^{3/8} * k^{-3/8} * J^{-3/16} \quad \text{and} \quad h = 0.1 * b \qquad [m] \qquad (6)$$

and $A = 0.1\, b^2$ $[m^2]$

(v) Flow velocity v

Finally, in order to determine the river section relevant to exposure L_{Expos}, the flow velocity v_{fl} must still be calculated according to equation (1).

$$v_{fl} = Q / A$$

4.6.4 Drift deposition frequency onto the body of water

On a river course of length *L* (e.g. on a scale of several kilometers), pesticide-treated fields border on the water body with a given percentage of 0 - 100 % (200 % when taking into account land on both sides). Within a time span *T* (e.g. during a common 8-hour workday), every field will be treated with pesticides in a stochastic succession.

A water unit which flows in a river alongside pesticide-treated fields can be affected (within the given river stretch *L*) by a spray drift deposit once, several times, or even not at all. The probability of how often a single water unit along its stretch of river encounters a deposit from a neighbouring treated field can be understood as a stochastic process. The probability that 0, 1, 2, 3, … spray drift deposits contaminate a single water unit can be described with a binomial distribution.

The binomial distribution describes the outcome probability of a series of similar experiments which only have two possible results (i.e. spray drift deposition “TRUE” or “FALSE”). If the desired experimental result (i.e. deposition “TRUE”) has the probability *p*, and the number of experiments is *n*, then the binomial distribution will specify how probable the total *k* results (with the attribute “TRUE”) will appear.

$$P(k) = \binom{n}{k} p^k (1 - p)^{n-k} \tag{7}$$

whereas:

P(k) Probability that a water unit encounters k = 0, 1, 2, … “incidents,” i.e. spray drift deposits, while passing *n* field sections

p Incidence rate of the incidents (i.e. spray drift deposits) during a single experiment, or while the water unit passes one treated field section

n Number of experiments, i.e. the total number of field sections (with the same length as the water unit) which the water unit passes.

<u>Example:</u> A 25 m long water unit flows through a river section 1 km in length. Along this river section there are n = 30 field sections of 25 m length each to be treated (i.e. 30 “experiments”). The probability that a field section will be handled during the exact time interval in which the water unit is affected by the spray drift deposit might be p = 0.02 (2 %). The probability P(k) that this unit of water has encountered k = 0, 1, 2, … deposits by the end of the 1 km river section can be calculated according to equation (8).

Table 4-20: Probability and cumulative probability that a water packet (of 25 m length) receives 0, 1, 2, ... drift deposition along its passage through a river stretch of 1 km (and 2 h flow duration) when 30 fields are treated randomly within a time-frame of 2.5 h.

Probability function f(x)	Cumulative Distribution function F(x)
P(0) = 0.545484	F(X = 0) = 0.545484
P(1) = 0.333970	$F(X \leq 1)$ = 0.879454
P(2) = 0.098828	$F(X \leq 2)$ = 0.978282
P(3) = 0.018824	$F(X \leq 3)$ = 0.997107
P(4) = 0.002593	$F(X \leq 4)$ = 0.999700
P(5) = 0.000275	$F(X \leq 5)$ = 0.999975
P(6) = 0.000023	$F(X \leq 6)$ = 0.999998
P(7) = 0.000002	$F(X \leq 7)$ = 1.000000
$P(\geq 8)$ = 0	$F(X \leq 8)$ = 1.000000

This means that the water unit has a 54.5 % chance of encountering no deposits, a 33.4 % chance of encountering exactly one deposit, etc.

The binomial distribution contains the expectation value E(X) with:

$$E(X) = \Sigma_{i=1}^{n} x_i p_i = np \tag{8}$$

In this example, the expectation value is n*p = 30 * 0.02 = 0.6, which means that a water unit in the middle of the river section will encounter 0.6 applications. Interpreted another way, 60 % of the water units will encounter a spray drift deposit (if one considers the deposits to be indivisible quantities).

In Table 4-21 the resulting expectancy values n*p are depicted for several combinations of *n* and *p* as well as the number of river segments which encounter *more* than one spray drift deposit. Figure 4-18 shows the progression of the dispersal function $P(X \leq k)$ for some of the combinations of *n* and *p* from Table 4-21.

The results indicate that when *n* increases, i.e. when there is an increasing number of treatments along a river section, there is an increased probability that an individual river segment will encounter deposits (the conclusion is initially trivial). However, there is an *above average* increase in probability that an individual segment will be subjected to spray drift *multiple times* (≥ twice).

Table 4-21: Expectation value and percentage of river segments which receive *more* than one spray drift deposit for a binomial distribution with various combinations of the parameter *n* (number of treated field sections) and p_i (probability that a river segment will encounter treatment n_i). In Figure 4-18 the distribution function graphs for several combinations are shown.

Number of treated field sections#	Incidence rate of individual deposit incidents	Expectation value E(X) = n*p	Percentage P(k) of river segments which expe-rience >1 deposit	Graph in Figure 4-18
n = 5	p = 0.025	0.125	0.6 %	(a)
n = 10	p = 0.025	0.25	2.5 %	(b)
n = 20	p = 0.025	0.5	8.8 %	(c)
n = 40	p = 0.025	1.0	26.4 %	(d)
n = 10	p = 0.1	1.0	26.4 %	--
n = 40	p = 0.01	0.4	6.1 %	(e)
n = 40	p = 0.005	0.2	1.7 %	--
n = 40	p = 0.001	0.04	0.1 %	(f)

#: along a river section with 40 segments

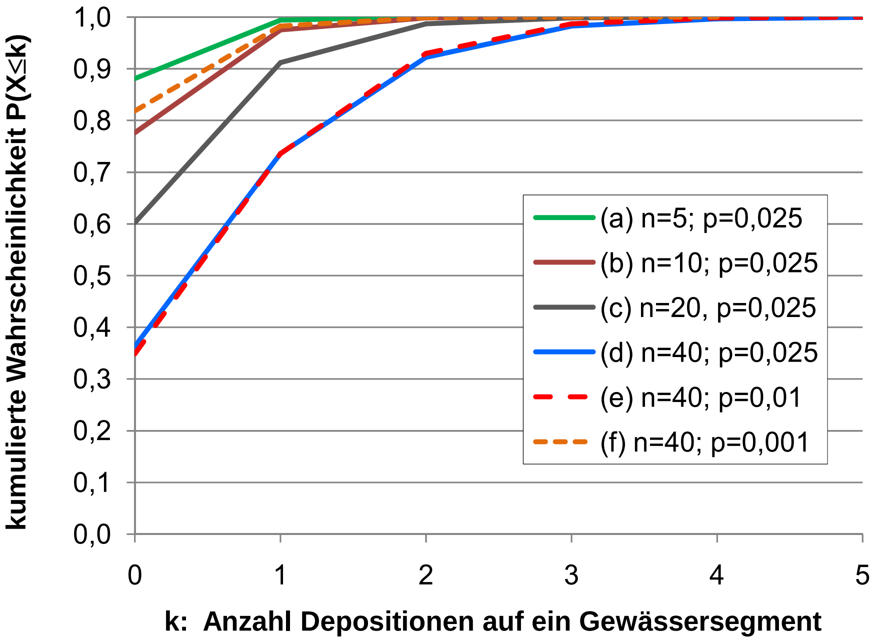

Figure 4-18: Cumulative distribution functions P(X ≤ k, k = number of deposits on a river segment) of the probability that a river segment of 25 m length within a section of the parameter *n* (number of treated field sections) and p_i (probability that a river segment will encounter treatment n_i)

4.6.5 Hydrodynamic dispersion

From the pesticide deposits in the water volume of the affected river segment a concentration impulse (initial concentration, $PEC_{initial}$) occurs first, corresponding to the dilution of pesticide deposits in the water volume. Due to *hydrodynamic dispersion* (longitudinal dispersion in the direction of flow), this originally compact particle-package will fan out into a steadily widening “cloud,” and the concentration peak will level out.

After a certain distance from the deposit segment *i*, the transport or rather the change in concentration of dissolved substances in a one-dimensional current with constant water flow can be expressed in a one-dimensional convection-dispersion-equation (cf. Fischer *et al.* 1979).

$$\delta C / \delta t = D_L (\delta C / \delta x^2) - v_{fl} (\delta C / \delta x) - k * C \qquad (9)$$

whereas:

C	Concentration	[µg/L]
x	Section of flow (with the current)	[m]
t	Time	[s]
v_{fl}	Mean flow velocity (with the current)	[m/s]
D_L	Longitudinal dispersion coefficient	[m²/s]
k	Reduction coefficient	[1/s]

The term δC/δt describes the temporal change in concentration caused by the inflow and outflow of particles in the measured volume element at any given time. The first term on the right side of the equation describes the hydrodynamic dispersion in the direction *x*, and the second term accounts for the convective transport. The analytical solution to equation (9) (for a conservative substance) results in:

$$C(x, t) = M/A*(1/\sqrt{(4\pi D_1 t)}) \exp[-(x - v_{fl}t)^2/(4D_1 t)] \qquad (10)$$

whereas:

C(x,t)	Concentration at location *x* at time *t*	[µg/L]
M	Substance mass	[mg]
A	Flow cross-section	[m²]
x	Flow section, distance from entry point	[m]
t	Flow time from the moment of entry	[s]

In the literature, many approximation formulas are given for the longitudinal dispersion coefficient many of which refer back to Fischer (1975).

$$D_L = 0.011\, v_{fl}^2 b^2/(hv') \quad [m^2/s] \qquad (11)$$

with $v' = (g\, R\, J)^{0.5}$ [m/s]

whereas:

v’ Shearing stress [m/s]

g Gravitational acceleration [m/s²]

According to equation (10), the *spatial* distribution of pesticide concentration with the current at a given location *x* in the river matches a normal distribution rate. Figure 4-19 shows the drop in initial concentration, schematizing a growing distance from the place of deposit.

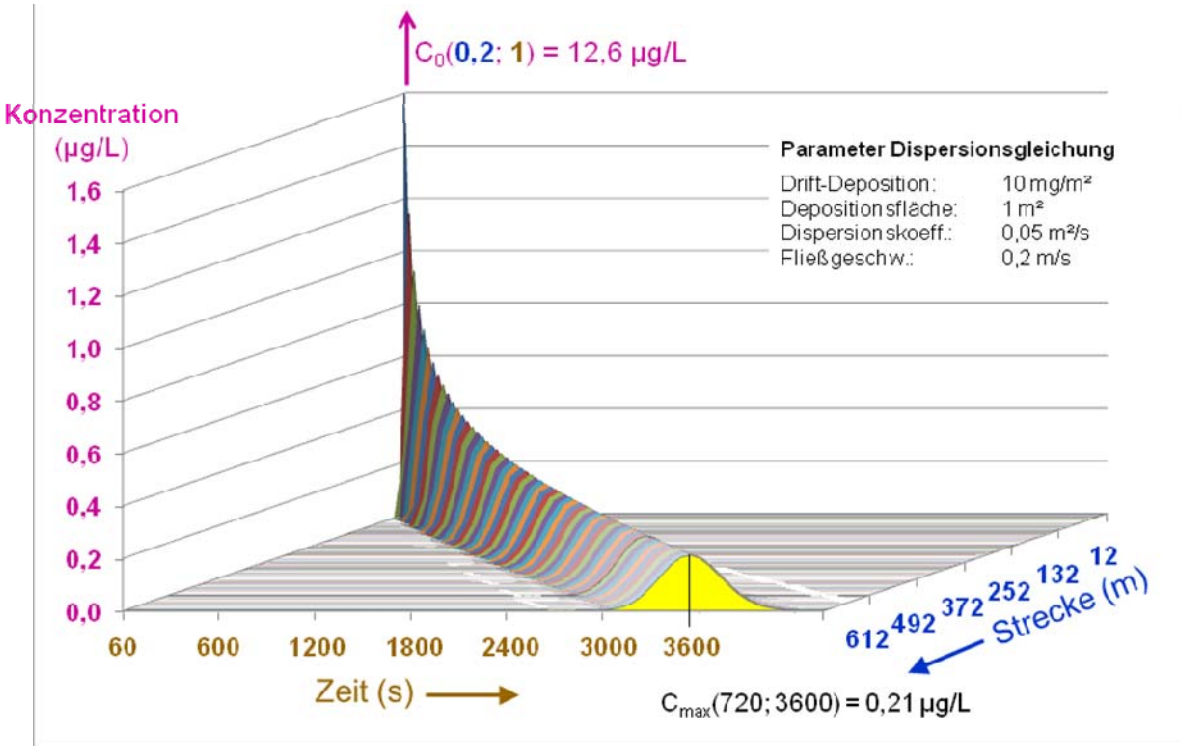

Figure 4-19: A schematized time-distance-profile of the substance concentration (after a pulse substance intake), for the range time (Zeit) t ≥ 60 [s] and distance (Strecke) x ≥ 12 [m] from the point of entry (calculated according to equation 8)

In streaming waters (smaller creek reaches) with “common” channel hydraulics, the initial concentration of a substance is quickly and measurably diluted after a relatively short flow section; thus the concentration in many rivers declines after about 100 - 200 m of flow to $1/10^{th}$ of the initial value or lower, and after 1 km of flow only a small percentage remains. However, for this reason, the substance cloud gets even more “dragged out” in streaming bodies of water. For a stationary organism in the water this means that the *duration* of exposure is respectively longer.

4.6.6 Exposure and effects

“Exposure” is normally understood as the time *T* during which an aquatic organism is exposed to the effects of a substance with a concentration *C* in the water. For a stagnant water body this concentration is $PEC_{Standard}$, a temporal and localized **constant** (provided that degradation processes have no part), the determination of which is trivial with known deposit amounts and water volumes. Moreover, this means that there is no difference in exposure in stagnant water for either sessile (stationary) or mobile organisms.

For a streaming river into which substances are discontinuously introduced, “exposure” becomes a **dynamic** factor: concentration *C* is a temporal and localized variable function C(x,t). This raises two key questions:

(a) “Exposure” is differentiated in order to account for sessile vs. mobile organisms.

For *sessile* organisms, the exposure matches the concentration $C(x_{fixed},T_{Expos})$ in the water flow, which passes their habitat x_{fixed} in the water during an ecotoxicologically relevant time period T_{Expos}.

For *mobile* organisms in the most simple case (i.e. when an organism drifts passively with the current), the exposure matches the concentration $C(L,T_{Expos})$, which adjusts itself in the water unit across the flow section *L* and which sets the water unit back during the time T_{Expos}.

(b) Under actual conditions one must assume that C(x,t)[2] is extremely variable. For practical reasons, this stochastic time-distance-profile of substance concentration in the flowing wave has to be averaged (smoothed) and transformed into a rectangular profile in order to be approved. The concentration profile will be converted into a rectangular profile $C_{Equ}(x,t)$, the area of which, according to the probability density function (cumulative frequency), is equal to the integral across the probability density function of the time-variant concentration.

In order to guarantee the protectivity of a PRA (Pesticide Risk Assessment) it must be required that the *maximum* concentration $C_{max}(\mu_s,t)$ of the time-variant concentration profile is maintained. For a normally dispersed concentration gradient, a length with an approximately ±1.25σ standard deviation from normal distribution for the equivalent rectangle profile (see Figure 4-20) arises from this requirement.

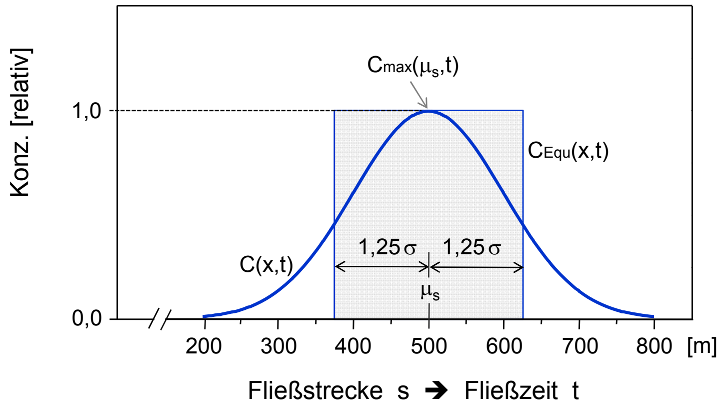

Figure 4-20: Transformation of a normally dispersed concentration function C(x,t) (in the example: with a maximum concentration of μs = 500 m and a standard deviation σ = 100 m) in an equivalent TWA concentration $C_{Equ}(x,t)$ overlength L_{Equ} and with the same maximal concentration $C_{max}(\mu s,t)$ and the same cumulative frequency (surface area under the curve).

However, considering the evaluation of effects (ecotoxicological studies) for this kind of equivalence function, the *relevant duration of exposure* T_{Expos} for organisms must be *specified* so that $C_{Equ}(x_{fest},T_{Expos})$ can be determined (this also applies to the exposure function $C_{Equ}(L,T_{Expos})$ of mobile organisms). Furthermore, it needs to be checked whether - from an ecotoxicological point of view - it is sufficient to operate with a mean concentration value all the time, or whether it may be taken into account the effects of some substances without the *maximal values*, which may be reached, for example, for a few minutes or an hour. Chapter 6 explains in detail what

[2] The notation C(x,t) here generally means “concentration as a function of time and place” and is not to be understood as an analytical solution as shown in equation (8).

consequences the exposure model has for streaming waters and especially how to implement an ecotoxicological assessment based on short-term exposure of organisms to pesticide substances.

4.6.7 Model concept: Concentration gradient in streaming waters in the general case

From the preceding explanations the following requirements must be conveyed for an exposure model that describes the time-distance-profile C(x,t) of pesticide concentration in a flowing body of water in general cases:

An approach (model) will be sought with which the concentration time response C(x,t) can be described for each river segment, 0,…,n, as a function of the application's occurrence in headwaters (as well as the given hydraulic water characteristics), see Figure 4-21.

For the registration decision the concrete function gradient C(x,t) at each location is not of interest; but of importance is the question on how often, or rather how long a certain critical (ecotoxicologically relevant) concentration threshold is exceeded. In other words, for each segment the distribution function F[C(t)] must be described.

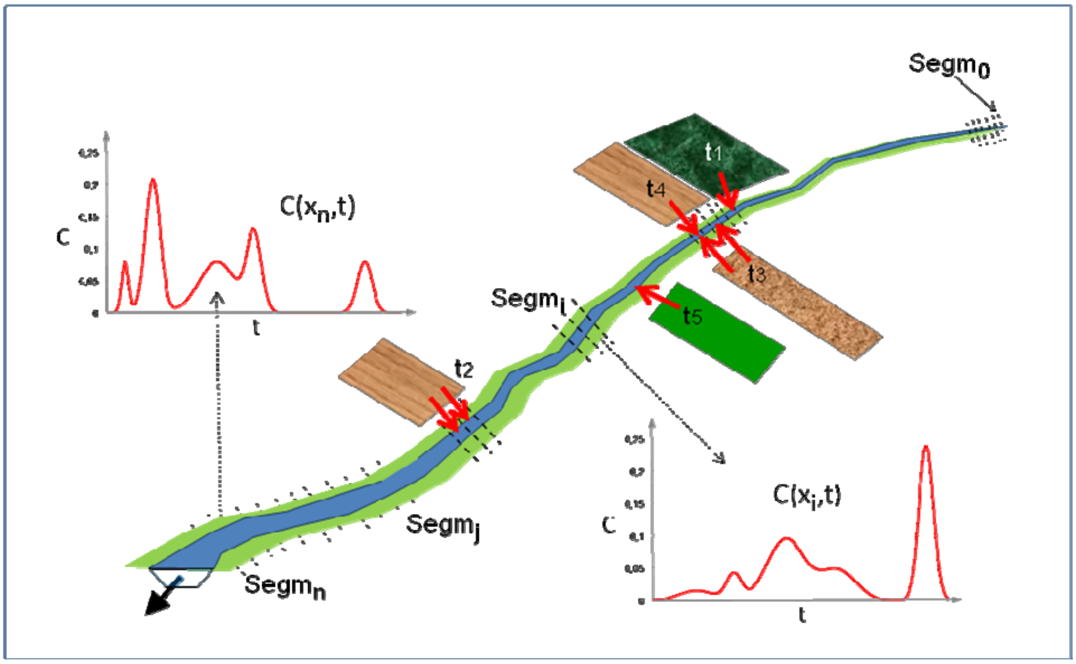

Figure 4-21: Conceptual schematic: pesticide treatment along a flowing body of water at various times t_1, t_2, etc., and the resulting time-distance concentration profile in various river segments $Segm_j$ in general cases. Arrows indicate pesticide deposition at time t_n

The concept of a dynamic exposure model for the special case of an *unbranched* streaming water with a *stationary* discharge and flow velocity is briefly described in the next chapter. However in general cases, branched river systems and/or courses with an *increasing drainage volume* have to be considered over the course of the flow section.

Furthermore, the structure of a river system as a branched tree must not be neglected in reality. Regarding converging water networks (branching streaming waters) the mean concentration C_{mean} after the confluence of two river branches 1 and 2 is given as

$$C = \frac{Q_1 \cdot C_1 + Q_2 \cdot C_2}{Q_1 + Q_2} \quad (12)$$

whereas:

C	Concentration [µg/L]
Q	Discharge [m³/s]
index 1, 2	River branch [1, 2]

For this problem two assumptions can be introduced for simplification: (i). The resulting concentration C_{mean} cannot exceed the highest of the concentrations C_1, C_2 of the two branches (the mathematical proof is simple, therefore we do not present it here). (ii) Typically a river network some kilometers downstream the source is characterized by a large main branch which receives small brooks from both sides. Hence, for the main branch holds $Q_1 >> Q_2$, and re-arranging equ. (12) leads to $C_{1mean} \approx C_1$. Therefore calculating the exposure modelling only the main branch of a river system is a simplified, but realistic approach.

Nevertheless, in a few cases the $PEC_{\_TWA(1h)}$ as well as the time over threshold could be higher after the confluence. But due to the fact that the flow path of the main branch is longer than that of the shorter branch - which probably also applies for the number of application fields and the pesticide input - this approach is an insignificant simplification. A detailed mathematical handling based on probabilistic calculations of the aspects mentioned above will be subject of further studies, but this is not an argument to disapprove the dynamic approach in total.

The parameterization of the dynamic model concerning the hydrological conditions based on ground truthing and in situ measures of flow velocities and water depths. As input for the model mean values of all single measures of the main branch of the ground truthing were used. This leads to an overestimation of the hydrological parameters in the head water, but also to an underestimation of flow velocities and water depths in the underflow. Due to the concept of calculating $_{TWA(1h)}$ and an adapted RAC with the time over threshold as an important input, the PEC dynamic increases following the flow path downstream (see Chapter 8). Therefore this can be considered as a conservative approach.

In Chapter 8.13 it is demonstrated as a practical example that effects of dilution are effective when smaller streams flow into water bodies with a significantly higher water volume and discharge. As a conclusion to this problem usually the highest concentration of pesticides is calculated for the main branch. Water depths and flow velocities are underestimated for the main branch after the confluence with a tributary. In so far the simplification can be considered a conservative assumption with respect to ecotoxicological risk assessment.

4.6.8 Exposure model for streaming water

As a first step, the concept for exposure modelling in streaming waters is outlined in the following section for the special case of *unbranched* streaming waters with *stationary* flow velocities (and other hydraulic parameters) and a simplified rectangular cross section.

The main idea behind the simplified method is to approximate the unknown (or rather undetectable) way-time concentration profile C(x,t) via transformation from C(x,t) into a mean PEC [mPEC(T_{Expos})], which averages all deposit entries within a defined observation time T_{Expos} in a defined river section (Figure 4-22).

The first step of this procedure for mPEC(T_{Expos}) requires specifying an exposure-relevant time span T_{Expos}, e.g. 1 h, 6 h, or 24 h. Whichever duration for T_{Expos} makes sense must take precedence and be established from an ecotoxicological (effect side) viewpoint, and it will have to be discussed further during the course of the project. For the upcoming discussion and calculations an exposure-relevant time span T_{Expos} of 1 h will be used first.

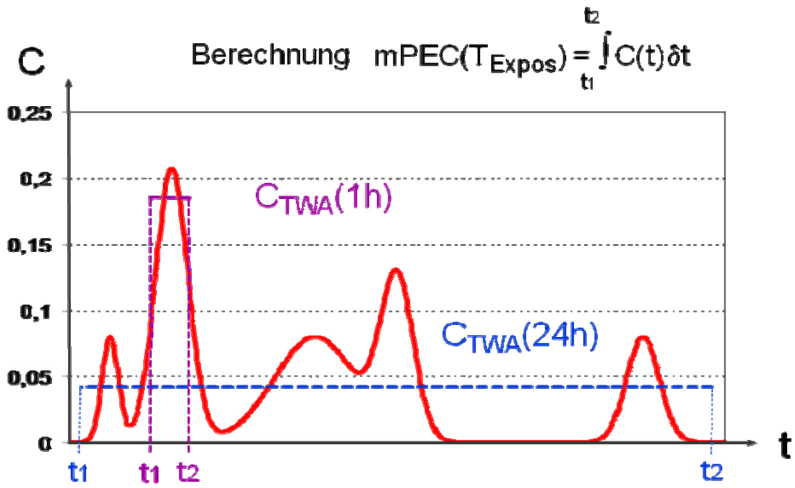

Figure 4-22: Schematic for the transformation of a (random) time-distance-concentration profile C(x,t) for an averaged PEC [mPEC(T_{Expos})] of a given duration T_{Expos}, here with a T_{Expos} of 1 h and 24 h, respectively

Note: Setting “1 h” as the exposure-relevant duration is presenting it conservatively. It can be assumed that the amount of mPEC(T) abates with an increasing *T* duration. An mPEC(48 h) is thereby (presumably) always *smaller* than mPEC(1 h). If mPEC(1 h) < RAC (Regulatory Acceptable Concentration; positive approval decision), then mPEC(48 h) < RAC.

The **exposure model** is based on the assumption that all drift deposits on the water volume (the bodies of water found moving in the river) within the exposure-relevant time span T_{Expos} mix in a spatially and temporally homogenous manner.

With T_{Expos}, via channel hydraulics, every exposure-relevant river section L_{Expos} for every river segment is also given whose spray drift deposit insertions determine the mPEC. Much more simply, the average concentration in bodies of water which passes a segment during T_{Expos} is in accordance with the substance load from all deposits which are inserted during time T_{Expos} into the water volume $A*L_{Expos}$ (upstream from the affected segment $Segm_j$) as described by Equation (12):

$$mPEC(T_{Expos})_j = \frac{\sum_{T_{Expos}} Load_i}{Q * T_{Expos}} = \frac{\sum_{Segm_{j-k}}^{Segm_j} Load_i}{A * L_{Expos}} \quad [\mu g/L] \qquad (12)$$

whereas:

T_{Expos}	Ecotox. relevant exposure duration (defined previously, here [1 h])
L_{Expos}	Flow section during exposure duration T_{Expos} [m]
mPEC(1h)	Spatially (via L_{Expos}) & temporally (via T_{Expos}) averaged constant [µg/L]
$Load_i$	Spray-drift deposit in Segment *i* during T_{Expos} or rather along L_{Expos} [mg] (stochastic process)

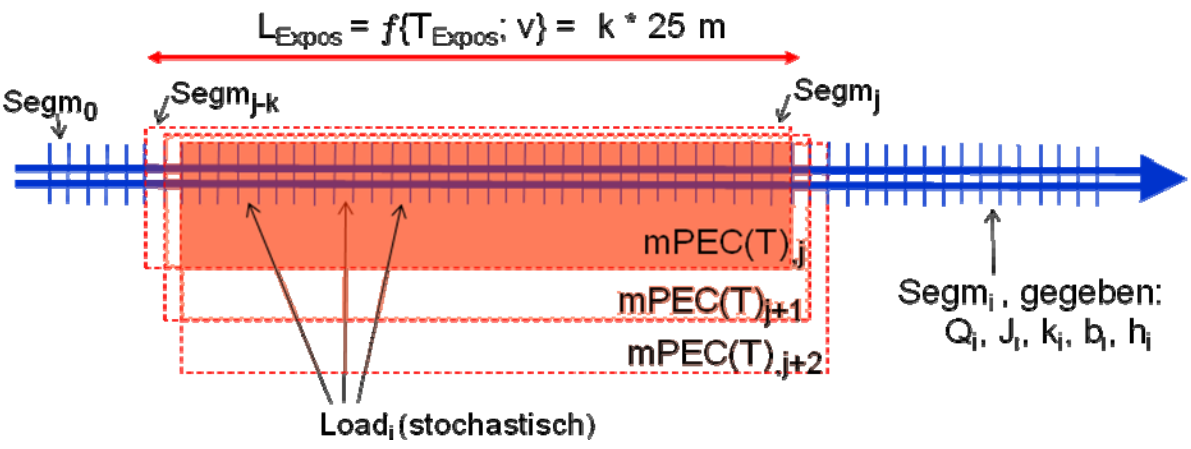

Figure 4-23**Fehler! Verweisquelle konnte nicht gefunden werden.**: Schematized scheme of a simplified exposure model for streaming waters: „moving window“ calculation of concentration mPEC(T), averaged over the duration T_{Expos} (= river stretch L_{Expos}), for river segments $Segm_j$, $Segm_{j+1}$, $Segm_{j+2}$ etc. The spray drift deposition $Load_i$ alongside L_{Expos} during T_{Expos} has to be modeled as a stochastic process.

The second term in equation (8) for the calculation of mPEC(T) needs geo-referenced data of spray drift depositions $Load_i$ along L_{Expos} during T_{Expos}. The $Load_i$ has to be treated as an stochastic process described by a binomial distribution (see Chapter 4.6.4). For an application for existing river stretches or hydrological regions informations on the following parameters are essential:

- Number or length of potentially treated fields alongside L_{Expos}
- Distance between surface water and edges of the treated fields (determines the spray drift loss)
- Number of fields which are really treated during T_{Expos}

In general the listed parameters and information is available for the landscapes in Germany or can be obtained with reasonable effort. In case of missing data a parameter has to be substituted by a conservative estimation.

An exemplary application of the exposure model for streaming waters is presented here, its mathematical concept and its implementation in a GIS environment is elaborated in details in Chapter 8 for streams in the hops growing region Hallertau.

4.7 Calculation of sediment concentrations

Substances characterised by a high sorption to soil disappear from the water phase and accumulate in sediment within a short time. Though the process has no impact on initial concentrations in the water, time dependent concentrations are significantly influenced. Presently this process is considered in the registration procedure by considering the disappearance from the water phase for long-term concentrations.

The calculation of the distribution of pesticides from water into the sediment is more complex than for instance the calculation of degradation in water or volatilisation from water. This is due to the fact that distribution into sediment is an equilibrium reaction. The system always tries to reach a fixed concentration ratio between water and sediment. The equilibrium can be calculated based on the sorption coefficient Kd or (if not available) based on the organic carbon content and the K_{oc} (sorption coefficient normalized to organic carbon content). Further necessary parameters are the depth of water and sediment layer and the density of sediment.

Of course a constant fraction (calculated on the basis of equilibrium conditions) could be relocated into the sediment during application (or at least 1 hour later to maintain the worst case character of the initial concentrations). However, this is in conflict with reality as due to different degradation rates in water and sediment the equilibrium is permanently disturbed and the system reacts with further relocation. Some fate models use a constant time step (1 day or 1 hour) to reach the equilibrium, whereas others calculate sediment concentrations based on concentration gradients according to Fick's law.

Though the fate models are more or less sophisticated to simulate this process always numerical (not analytical) procedures must be used because different transport and degradation processes have to be considered in parallel. It is questionable whether these methods can be implemented into the current geo-referenced system.

A possible solution would be to consider the calculation of time dependent concentrations in water and sediment by using a fast fate model like STEPS (Klein 2007) outside the geo-referenced software. This could be realised based on a special interface between model and GIS-software to provide the fate model with all necessary information of the local surface water (e.g. depth, width, stream flow). Similarly, the fate model could report back to the GIS-software based on a special interface summarising the results of the simulation.

An alternative to extensive modelling would be to calculate conservative maximum sediment concentrations in the system only, without simulating time dependent concentrations and without considering the effect of sediment concentrations on the concentrations in the water phase. The results of a water-sediment-study could be used to estimate respective concentrations.

4.7.1 Availability of the input parameters and models

For the simulation of the relocation of pesticides into the sediment first of all equilibrium conditions between water and sediment have to be known. This can be done based on the K_{oc}-value, for which average values are publicly available for all pesticides. However, also sediment depth, sediment density and the organic carbon content in sediment must be available, which is not generally the case. As long as these parameters are not available on a local level standard parameters (e.g. sediment depth: 5 cm, organic carbon content in sediment: 5 %, sediment density: 1.3 g/cm³) or those in FOCUS-surface water (FOCUS 2002) could be considered.

Time dependent concentrations in water and sediment could be generally simulated using the SWASH shell which combines in total 3 different models (PRZM, MACRO and TOXSWA) as explained in the FOCUS-SW-report (FOCUS 2002). However, as shown by Klein (Klein 2007) the model STEPS-1-2-3-4 has some advantage compared to TOXSWA (e.g. it runs significantly faster) and therefore has also been used within the FOOTPRINT project instead of TOXSWA. As the concentrations calculated by both models are practically identical, STEPS was used to demonstrate the effect of the process relocation into sediment. Both models consider predefined surface water scenarios as developed by FOCUS 2002 for their calculations, but the models generally could also handle scenario parameters from other sources (e.g. from the new geo-referenced probabilistic model, if a suitable interface has been established). Unfortunately, the FOCUS scenarios do not allow switching off run-off entries when using the models because for registration processes the effect caused by runoff and drift events are always summed up. Therefore, the time dependent concentrations presented in the following figures are always caused by both entry routes.

4.7.2 Sensitivity of the process "distribution into sediment"

Generally, this process will not influence the initial concentrations. Only time dependent concentrations are affected. In the following two figures time dependent concentrations are presented for the scenario European FOCUS scenario R1 (pond in Southern Germany) for an autumn application in winter cereals considering a weakly sorbing pesticide (K_{oc} 15 L/kg, Figure 4-24) and a strongly sorbing pesticide (K_{oc}: 5000 L/kg, Figure 4-25). Obviously, totally different concentrations are simulated for the two compounds. This is caused by run-off rather than spray drift because pesticide surface run-off and erosion losses from fields are very sensitive to sorption in soil. However, the model demonstrates that distribution into sediment is not a dominant process in this pond scenario.

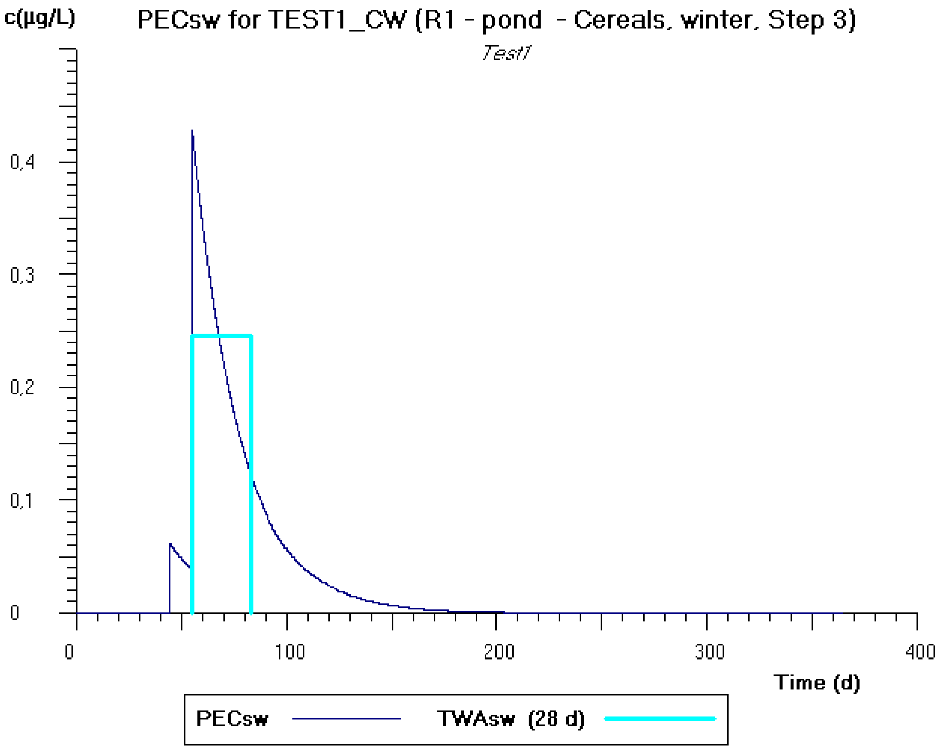

Figure 4-24: PEC_{sw} for a weakly sorbing pesticide (K_{OC}: 15 L/kg) when applied in winter cereals in autumn

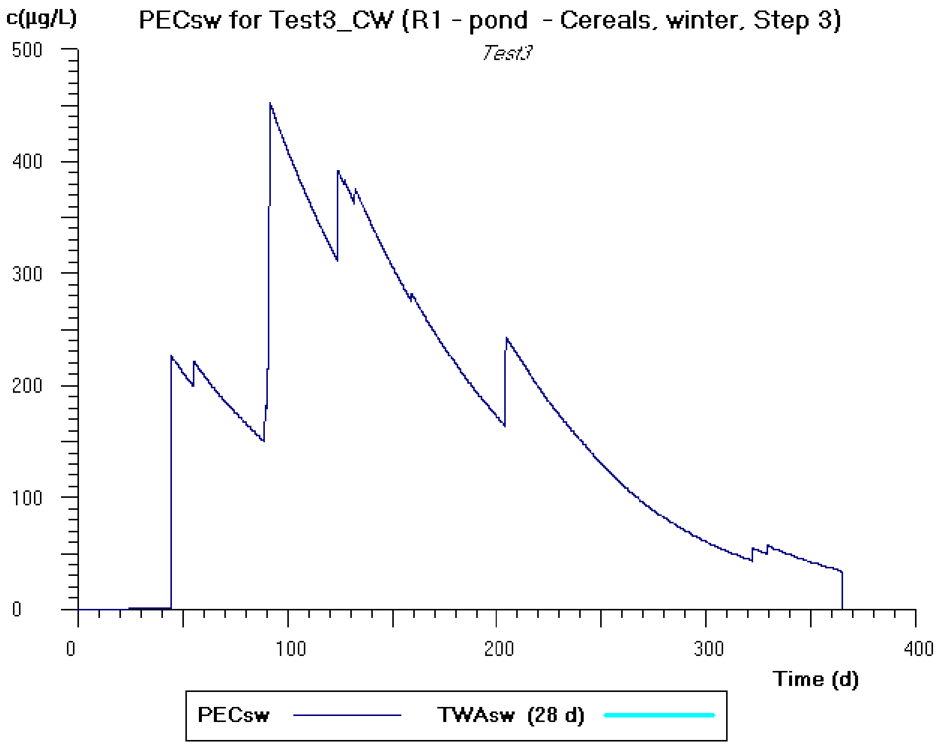

Figure 4-25: PEC_{sw} for a strongly sorbing pesticide (K_{OC}: 5000 L/kg) when applied in winter cereals in autumn

The following two figures demonstrate that also distribution into the sediment phase is totally different for the two compounds. Obviously, relocation into the sediment can be neglected for low sorbing compounds for which the water phase is the dominant medium. Accumulation is not expected for these substances because amounts that may have temporarily diffused into the sediment would leave the sediment phase as soon as permanent input and outflow in the surface water body have reduced the pesticide concentration there. Consequently, the decline shown in Figure 4-26 is not caused by degradation but by re-location from the sediment phase back into the water phase.

Significantly higher concentrations can be expected for strongly sorbing compounds as shown in Figure 4-27. Relocation into the water phase will take more time.

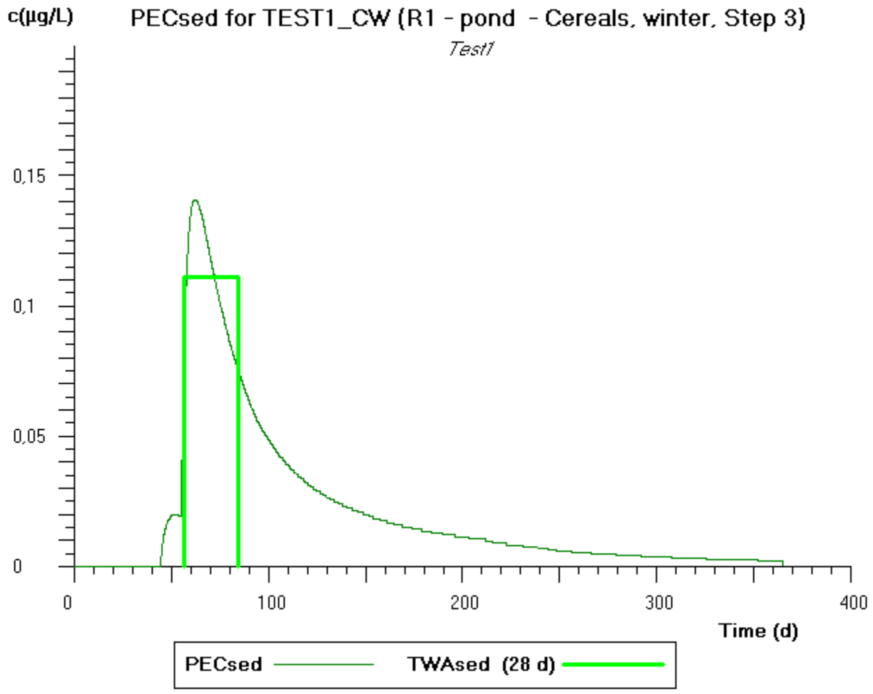

Figure 4-26: PEC_{sed} for a weakly sorbing pesticide (K_{OC}: 15 L/kg) when applied in winter cereals in autumn

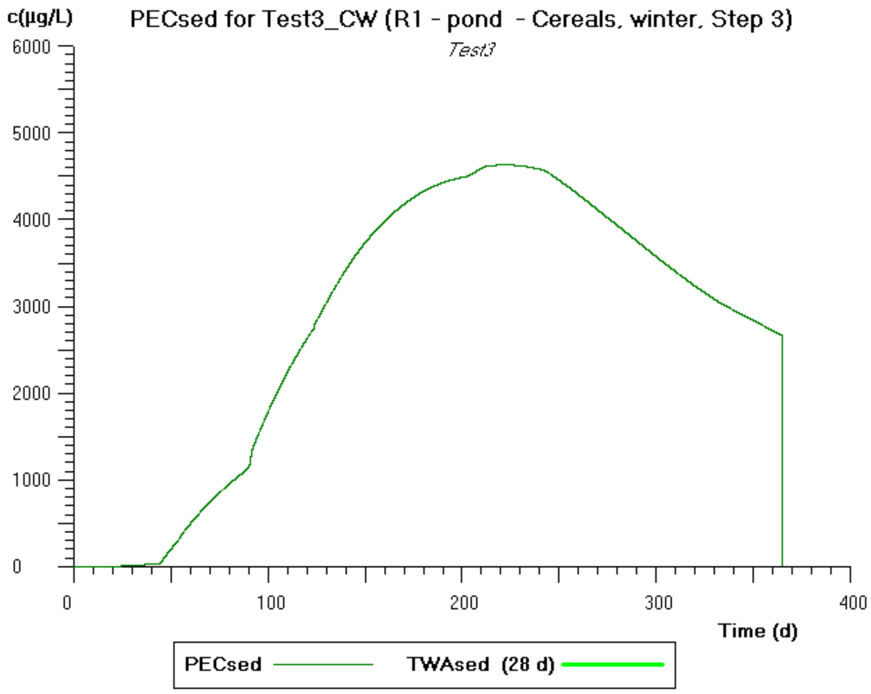

Figure 4-27: PEC_{sed} for a strongly sorbing pesticide (K_{OC}: 5000 L/kg) when applied in winter cereals in autumn

Alternatively, the estimations of maximum occurance in the sediment phase can be done without doing complex fate simulations taking into account the results of water-sediment studies according to the following equation:

$$PEC_{SEDIMENT} = \frac{PEC_{SURFACE\,WATER} \cdot F_{\max}}{DENS_{SEDIMENT} \cdot 100}$$

$PEC_{SEDIMENT}$ local sediment concentration [µg/kg]

$PEC_{SURFACEWATER}$ local surface water concentration [µg/L]

F_{max} maximum occurance in the sediment phase [%]

$DENS_{SEDIMENT}$ sediment density = 1.3 [kg/L] (TGD, 2003) or measured value

An example for typical results of a water sediment study is presented in Table 4-22. $PEC_{SEDIMENT}$ is calculated based on the maximum occurrence in the sediment phase (in the example presented in Table 4-22: 57.1 %) together with $PEC_{SURFACEWATER}$.

Table 4-22: Results of a hypothetical water-sediment study

Time (d)	Substance fraction in the water phase (%)	Substance fraction in the sediment phase (%)
0	83.5	8.4
1	76.3	11.2
3	51.5	26.9
7	28.3	54.5
16	13.6	47.3
29	2.6	57.1
61	1.6	46.6
110	0.7	36.2
157	0.2	34.6

The method fits well for static water bodies. For the estimation for streaming water bodies a suitable substance fraction would be the occurrence at the first measurement (in the example above: 8.4 %).

The calculation can be performed also for the main metabolites. Only an additional correction considering the ratio of the molecular masses of parent and metabolite has to be made.

4.7.2.1 Conclusions: sediment concentrations

For the generic examination distribution into sediment cannot be considered, as this process is highly substance specific.

However, according to the model calculations reduction rates caused by distribution into the sediment layer can be a significant process for substances showing strong sorption to sediment.

Even if this process is relevant for specific compounds it will not be possible to consider this process directly in the new geo-referenced system because calculating the distribution of pesticides from water into the sediment is more complex than for instance the calculation of degradation in water or volatilisation from water. This is usually done based on special interfaces between the geo-referenced model and fate models and before this process could be considered as an additional step outside the geo-referenced system. However, within the FOOTPRINT project it was demonstrated that the STEPS algorithm can be integrated directly into the GIS-environment.

Nevertheless, it is recommended to use the simple estimation procedure based on the results of a water-sediment-study for $PEC_{SEDIMENT}$ instead, as long as spray drift is the only entry route.

This evaluation may change if runoff and drainage entries will be considered as additional input into surface water in the system because they highly depend on the sorption behaviour of the compounds.

4.8 Summary of the assumptions for the PEC-estimations in GeoRisk

In the preceding chapters the basic concepts of the GeoRisk approach are described in detail. As part of this approach a large number of variables and parameters are introduced in the context of the drift deposition calculation, the exposure assessment for static and dynamic water bodies, the evaluation of ecotoxicological effects, the risk mitigation measures, and finally the identification of potential management segments. Table 4-23 gives a comprehensive overview over the most important parameters and variables, ruling the calculation of the criteria for pesticide authorisation decision. The list allows a comparison of the variables with identical or similar factors used by other spray drift risk approaches (conventional approach; FOCUS) and to judge the maybe specific methodology of its derivation. Furthermore, the parameters are characterised with respect to their stochasticity (probabilistic vs. deterministic), their degree of protectivity, and their suitability to serve as a geo-referenced variable.

Table 4-23: Comprehensive overview on parameters and variables used for exposure estimations

Variable, parameter	**Probabilistic/ deterministic[a]**	**Protectivity, percentile[b]**	**Geo-referenced[c]**	**Methodology, value(s)**	**Comments**
Drift deposition calculation					
Wind direction	d	yes[b]	no	Distance analysis for n = 8 directions	ref. Chap. 4.2
Wind speed	---	yes[b]	---	Variability of wind speed is not considered explicitly (but drift deposition trial data covers a range of wind speeds)	ref. Chap. 4.2
Distance edge of field – edge of water body	d	yes[b]	yes	GIS analysis for 8 directions, reference point: center of river segment; Protectivity: distance measured from edge of bank (not edge of water surface)	ref. Chap. 4.2, 5.3, 5.4, 8.4 ATKIS- or HR-based

Variable, parameter	Probabilistic/ deterministic[a]	Protectivity, percentile[b]	Geo-referenced[c]	Methodology, value(s)	Comments
Buffer size of spray drift deposition from fields along water	d	yes[b]	yes	Buffer width: 150 m	ref. Chap. 5.3
Deposition rate	p	yes[b]	yes	Distribution function of spray drift deposition, recalculated by MC from JKI original field trial data. Geo-referenced with respect to the variable “distance edge of field – water body”	ref. Chap. 4.2
Deposition indicator for an individual water body segment	d	90-P	yes	Percentile of the deposition distribution influence by “distance edge of field – water body”, deposition rate associated to wind direction, deposition rate, Drift reduction by shielding waterside vegetation	
Drift reducing sprayer technique	d	yes[b]	no	Fixed factors 75, 90 %	ref. Chap. 4.4
Drift reduction by shielding waterside vegetation (hedges, windbreak)	d	yes[b]	yes	Fixed factor 25 % for MS identification. Protectivity: reduction factor is higher during summer and autumn For registration purposes reduction during the year is expressed with a trapeze function	ref. Chap. 4.4, 5.4
Drift reduction by emerse vegetation and shielding herbs	---	yes[b]	---	Not considered Protectivity: smaller brooks with low flow velocity are often (at least partly) covered by emerse vegetation during summer and autumn	ref. Chap. 4.4
Deposition from volatilization	---	?	---	Not considered for generic risk assessment (depends highly on substance vapor pressure)	ref. Chap. 4.2

Variable, parameter	Probabilistic/ deterministic[a]	Protectivity, percentile[b]	Geo-referenced[c]	Methodology, value(s)	Comments
Affected water bodies					
Location of water bodies	d	no[d]	yes	ATKIS analysis; HR analysis	ref. Chap. 4.1, 5.1 and 8
Type of water body	d	?	yes	ATKIS Object Type a) Lentic: • lakes (near bank line), ponds • rivers (width of >12m, ATKIS object type 5101,5103. 5112) b) Lotic: • rivers, ditches	ref. Chap. 5.3, 5.4
River network system	d	?	yes	a) Static: ref. line "Location" b) Dynamic: Prerequisite "river network topologically correct, i.e. flow directed and all gaps are closed"	ref. Chap 5.3 and 8
Segmentation river/ditch/lake (bank line)	d	?	yes	Segment length: 25 m	ref. Chap. 5.3
Exposure assessment – static water model					
Receiving water volume - Stagnant ditches (ATKIS object type 5103 – stagnant ditch)	d	?	yes	1 m width, 0.15 m depth (width/depth ratio 6.6 : 1) [g, h]	w/d-ratios ref. Table 4-18
Receiving water volume - Near bank area of lakes and of ditches or rivers with a width of > 12m (ATKIS object type 5101, 5103, 5112)	d	?	yes	1 m width (affected part of a water body), 0.3 m depth	ref. Chap. 5.3
Volatilization from water surface	---	yes[b]	---	Not considered for generic risk assessment (depends highly on substance properties)	ref. Chap. 5.3
Sediment concentration	---	?	---	sorption/desorption processes not considered (not relevant for short term exposure assessment)	ref. Chap. 5.3
Risk indicator	d	yes[b]	yes	$PEC_{initial}$ = Depos. / Water volume for individual segment; protectivity results from deposition calculation	

Variable, parameter	Probabilistic/ deterministic[a]	Protectivity, percentile[b]	Geo-referenced[c]	Methodology, value(s)	Comments
Exposure assessment – dynamic water model					
Receiving water volume – Flowing watercourses	p	mean	yes	Function of hydraulic features of water body and pesticide treatment parameters (both listed below).	ref. Chap. 4.6
River discharge	d	mean	yes	Primary data: catchment water balance and discharge model (pre-cipitation data, GIS analysis)	ref. Chap. 4.6.3
River bed slope gradient	d	mean	yes	Primary data: DEM analysis	ref. Chap. 4.6.3
Mannings roughness coeffi-cient	d	mean (?)	no	Primary data: literature value (15 $m^{1/3}$ s^{-1})	ref. Chap. 4.6.3
River depth	d	mean	yes	Calculated acc. Manning-Strickler equ.	ref. Chap. 4.6.3
River width	d	mean	yes	Calculated acc. Manning-Strickler equ.	ref. Chap. 4.6.3
Flow velocity	d	mean	yes	Calculated acc. Manning-Strickler equ.	ref. Chap. 4.6.3
Dispersion coefficient	d	mean	yes	Calculated acc. Fischer et al. (1979)	ref. Chap. 4.6.5
Superposition of drift deposi-tions	p	mean	yes	Binomial distribution: probability, that a flowing water package re-ceives n=0, 1, 2, … drift depositions	ref. Chap. 4.6.4
Treatment time frame for the application of all fields along river stretch	d	?	no	From NEPTUN 2006 the maximum percentage of fields treated at the same day with insecticides vary between regions from to 36 % (Re-gion Niederelbe) to 52 % (Region – Mitteldeutsches Obstanbaugebiet) (Golla & Rossberg 2010)	data on within day treatment time needed
Risk indicator	p	yes[b]	yes	$maxPEC_{TWA(1h)}$ = $\Sigma(\text{Depos.})_L$ / $\Sigma(\text{water discharge})_L$ with L=length of water course of flow time 1 h	ref. Chap. 4.6.6
Water body profile geometry	d	(yes)	no	Geometric form of river bed: rectan-gular (more conservative with re-spect to maxPECTWA than trape-coid)	ref. Chap. 4.4.3
Effect of tributaries	--	?	--	Not respected: superposition of $maxPEC_{TWA(1h)}$ at confluence and further downstream of two river stretches	ref. Chap. 4.6.7
Non-stationary hydrological conditions along river stretch	--	?	--	Not respected: river hydrology, especially $maxPEC_{TWA(1h)}$ constant over river stretch	ref. Chap. 4.6.7

Food notes

a) p: probabilistic distributed variable, d: deterministic variable;

b) degree of protection (conservation) of parameter estimation: percentile of value (in case that an exact determination of a percentile is possible or defined by the methodology of derivation)
Protection “yes” means: the value(s) is/are chosen beyond the mean or median of its distribution, but their degree of probability (an xy-percentile) cannot be identified exactly.

c) Geo-referenced variable: do values of the variable differ for spatial units (regions, river branches, river segments)

d) As long as it is not satisfied that all water bodies are captured by GIS analysis without any exception it is to assume that among the undetected water bodies are some segments which are "at risk".

e) in all permanent crop regions (vine, fruit trees, and hops)

f) Identification of potential management segments (Chapter 7 and 8) : Only these parameters and variables are mentioned in the Table whose values or assumptions differ from the 'standard' GeoRisk approach defined in the preceding chapters for the respective variable.

g) A mean ratio of 6.6 : 1 for static ditches in Germany is based only on a very small database on measured width/depth ratios for real water bodies. For later applications this values has to be refined and validated, see detailed discussion in chapter 4.4.1.2.

h) Please note: Differing to the w/d ratio given here, for the exemplary technical GIS implementation of the Hotspot identification (see chapter 5) the FOCUS scenario of 3.3 : 1 was used as an preliminary value.

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4.10 Appendix

Table 4-24: Statistical values of the deposition data for hops

		Lower wind speeds (<1.5 ms^{-1})				Higher wind speeds (>=1.5 ms^{-1})		
	xmin	N_x	Mean_x	Std_x		N_x	Mean_x	Std_x
Hops	3	9	12.69248	7.009425		10	9.516767	3.6781
	4	9	9.255074	5.825194		10	7.750767	1.92999
	5	9	7.218852	4.455982		10	6.6547	2.017588
	7.5	9	4.113481	3.033631		10	4.0825	1.685222
	10	10	2.525433	2.240013		11	3.215303	1.467388
	15	9	1.187407	1.273495		10	1.664367	1.167053
	20	10	0.514	0.524845		11	1.082303	0.957057
	30	9	0.206667	0.226359		10	0.371533	0.544669
	40	1	0.01002			8	0.0885	0.070701
	50	10	0.065545	0.057421		11	0.07161	0.085245

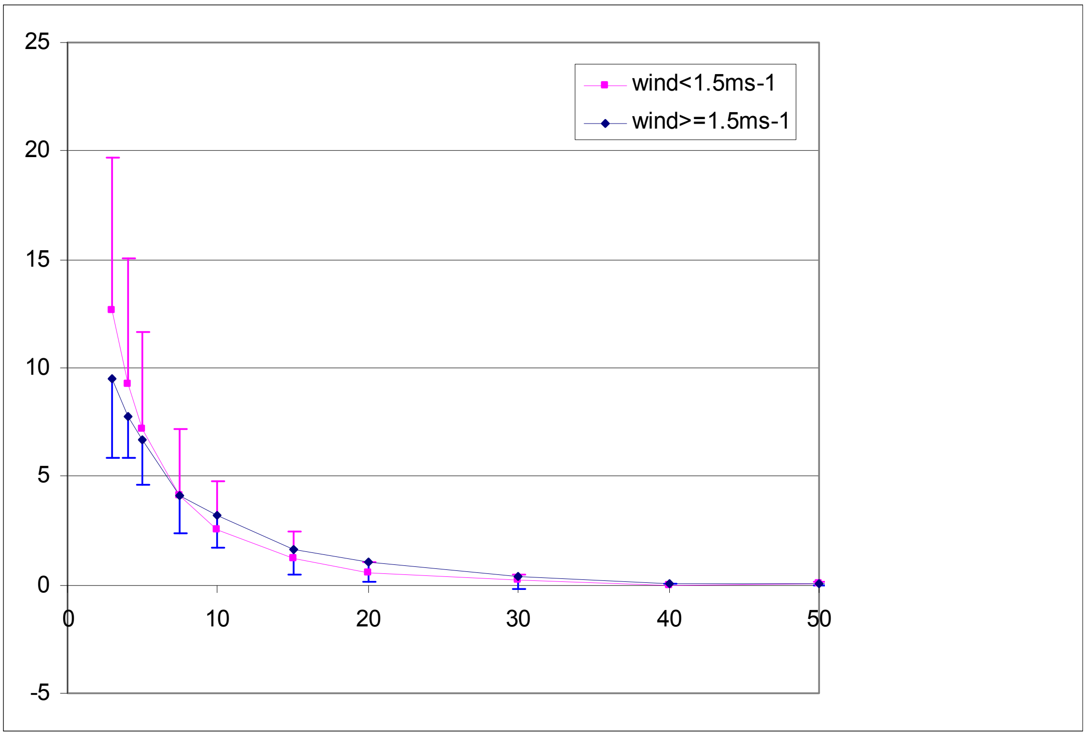


Figure 4-28: Means and std (one side) of trials per measurement distance and wind speed group of the deposition data for hops

Table 4-25: Statistical values of the deposition data for orchards

		Lower wind speeds (<1.5 ms^{-1})				Higher wind speeds (>=1.5 ms^{-1})		
	xmin	N_x	Mean_x	Std_x		N_x	Mean_x	Std_x
Orchard	3	4	11.7795	5.354369		26	19.42721	4.34702
	5	4	6.8455	3.680655		26	13.00909	3.02694
	10	4	1.924	1.118222		26	6.44492	1.951194
	15	4	0.6835	0.343104		26	3.291288	1.187211
	20	4	0.296875	0.104164		26	2.194302	0.90763
	30	4	0.094763	0.036414		26	1.050589	0.52963
	40	4	0.040411	0.033033		10	0.173425	0.101405
	50	4	0.021035	0.011064		10	0.080713	0.051933

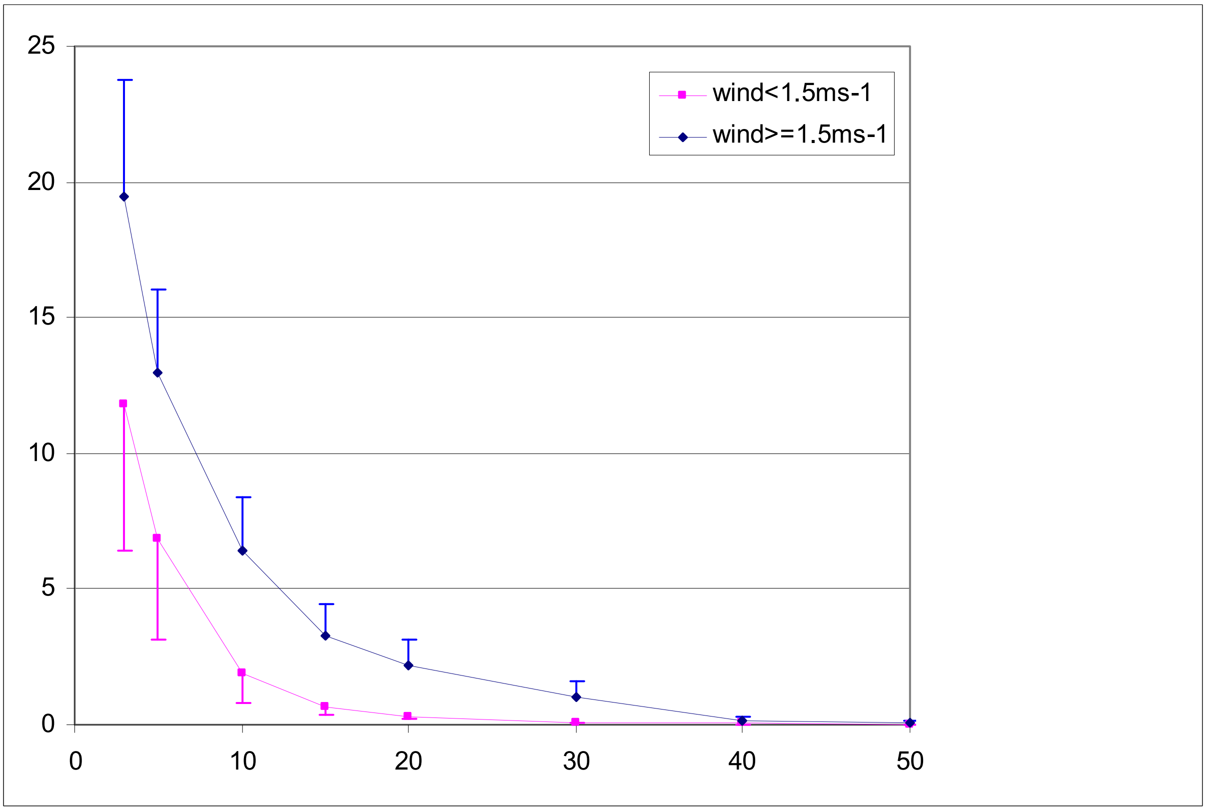


Figure 4-29: Means and std (one side) of trials per measurement distance and wind speed group of the deposition data for orchards

5 Concept realization with GIS

Burkhard Golla, Jens Krumpe

5.1 Summary

The tasks of the GeoRisk project include the technical implementation of a GeoRisk drift deposition modul and a GeoRisk exposure module for stagnant water bodies (GeoRisk static) according to the GeoRisk parameter definition (see chapter 4.1). Databases are built which contain the information required to conduct an exposure and risk assessment following the GeoRisk approach. Based on the outcomes a web-based software application is designed and implemented that allows stakeholders to conduct an exposure and risk assessment and to download spatial data for further refinements. The workflow for building up spatial domain databases for the GeoRisk approach is described in the first part of the technical document (see Appendix). In the second part, the system documentation and user manual of GeoRisk-WEB is presented. Relevant data sources and the uncertainty of data are explored, and a method for data integration is recommended (see chapter 5.4). The underlying workflow for creating a spatially explicit database containing the relevant landscape elements and generic exposure data is documented (see chapter 5.3 and Appendices). In order to assure the performance and practicability of the new approach, options for the upcoming data management at the UBA and personal expenses in the context of pesticide regulation are explored.

5.2 Introduction

The new geo-referenced probabilistic risk assessment approach GeoRisk is technically implemented according to GeoRisk parameters (see chapter 4). The descriptions and results presented in the following chapters focus on the GeoRisk approach for drift deposition simulation and exposure assessment for static water bodies (GeoRisk static). The exposure assessment approach for flowing waters is dealt with in chapter 8 (GeoRisk dynamic).

In this chapter the activities effecting the GeoRisk concept realization with GIS are grouped to the following topics:

Basic spatial domain database of GeoRisk: Chapter 5.3 describes the design of a spatial data model for the GeoRisk approach as a prerequisite for a uniform basic spatial database and displays the process of generating domain specific data.

Characterisation of uncertainty and errors: In chapter 5.4 the degree to which the underlying spatial data sets meet the requirements of GeoRisk is analysed and characterised.

Technical concept of GeoRisk: Chapter 5.5 describes the development of a concept for the management of domain and spatial data at the UBA in the framework of PPP regulation. This topic covers the aspects of data integration, data maintenance and cost estimation.

Technical Document: The technical documentation is a separate document (appendix B of this report. In the first part the steps are described that were taken to create spatial domain data based on ATKIS BDLM (Amtliches Topographisch-Kartographisches Informationssystem – Basis Digitales Landschaftsmodell, AdV 2003) for the GeoRisk approach. The intention is to enable UBA and other interested parties to build up such a database following a step-by-step procedure without depending on specific GIS software products (N). The second part of the document contains the system documentation of the GeoRisk application “GeoRisk-WEB”. The tools are meant to support the conduction of an exposure and risk assessment according to the GeoRisk approach. It describes the technical aspect of the application development and gives examples on how to use the application (L).

5.3 Basic spatial domain databases of GeoRisk

The GeoRisk approach requires a countrywide spatial database of all surface waters being theoretically exposed to pesticides spray drift from permanent crops (hops, vine and orchard). It requires a hydrologically correct topological relationship (connectivity) among water bodies and information on their distance to permanent crop sites as well as information on the presence of intervening drift filtering vegetation. For GeoRisk static detailed information on the surrounding landscape is necessary. Additional parameters are required for calculating the exposure of streams with GeoRisk dynamic.

One of the central goals of the project is the creation of a centralised and uniform basic spatial database (synonym: spatial data pool) which stores required the information to perform an ERA according to GeoRisk requirements. Comparable with a so-called uniform basic data set in health statistics it provides a set of minimum specifications for the content of information systems (Murnaghan 1978). Such data sets define the central core of data about a given fact needed on a routine basis by the majority of decision makers, and it establishes standard measurements, definitions and classifications for this core. Transferring this definition into the GeoRisk context a basic spatial database is specific for the domain of spatial ERA according to the GeoRisk approach. It stores the necessary data for calculating spatially explicit drift deposition distributions (GeoRisk Landscape Database, see chapter 5.3.4) and stores the aggregated results of the deposition simulation and derived exposure concentrations for GeoRisk static as input for hotspot computations (GeoRisk Exposure Database, see chapter 5.3.5). The parameter requirements for the GeoRisk dynamic model concept are more specific and described in chapter 8.

Table 5-1: Spatial parameter requirements for GeoRisk drift deposition assessment and GeoRisk static based on Table 4-23

Variable / Parameter	**Probabilistic/ deterministic** [a]	**Protectivity, percentile** [b]	**Geo-referenced** [c]	**Methodology, value(s), references**
GeoRisk drift deposition simulation model				
Wind direction	d	yes[b]	no	Distance analysis for n = 8 directions (0°,45°…315°); (Hendley 2001)
Wind speed	---	yes[b]	---	Variability of wind speed is not considered explicitly but drift deposition trial data cover a range of wind speeds within GAP (BBA 1992)

Variable / Parameter	Probabilistic/ deterministic [a]	Protectivity, percentile [b]	Geo-referenced [c]	Methodology, value(s), references
Distance edge of field – edge of water body	d	yes [b]	yes	GIS analysis for 8 directions, reference point: center of river segment for line features, edge of water surface for polygon features. Relevant distance for drift calculation: edge of bank (not edge of water surface) (Ganzelmeier et al. 1995)
Buffer size of spray drift deposition from fields along water bodies	d	yes [b]	yes	Buffer width: 150 m (Enzian & Golla 2006)
Deposition rate	p	yes [b]	yes	Distribution function of spray drift deposition, recalculated by MC from JKI original field trial data. Geo-referenced with respect to the variable “distance edge of field – water body” (EUFRAM 2006)
Deposition indicator for an individual water body segment	d	90-P	yes	Percentile of the deposition distribution influence by “distance edge of field – water body”, deposition rate associated to wind direction, deposition rate, drift reduction by shielding waterside vegetation
Drift reducing sprayer technique	d	yes [b]	no	Fixed factors 75, 90 %
Drift reduction by shielding waterside vegetation (hedges, windbreak)	d	yes [b]	yes	Weekly reduction rate during the year is expressed with a trapeze function.
Exposure assessment – static water model				
Receiving water volume - stagnant ditches (ATKIS object type 5103 – stagnant ditch)	d	?	yes	Calculation with a width/depth ratio (WDR) of 3.3 : 1, if not stated differently. Field data indicate higher WDR for ditches. More and regionalized data needed (see chapter 4.2). [d]
Receiving water volume - near bank area of lakes and of ditches or rivers	d	?	yes	1 m width (affected part of a water body), 0.3 m depth
Location of water bodies	d	no	yes	Spatial Database ATKIS-BDLM (AdV 2003)

Variable / Parameter	Probabilistic/ deterministic[a]	Protectivity, percentile[b]	Geo-referenced[c]	Methodology, value(s), references
Type of water body	d	yes	yes	ATKIS Object Type (AdV 2003) a) Static: - stagnant ditches - lakes (near bank) b) Dynamic: - rivers (streaming)
River network system	d		yes	a) Static: consistently (topologically correct) b) Dynamic: plus flow directed and closing of longer gaps (hydrologically correct)
Segmentation river/ditch/lake (bank line)	d		yes	Segment length: 25 m
Risk indicator	d	yes[b]	yes	$PEC_{initial}$ = Depos. / Water volume for individual segment; protectivity results from deposition calculation
Identification of potential Management Segments (MS) – static water model				
Hotspot-criterion for identification of MS	d	?	yes	Map 1 according to generic hotspot criterion: within a 1000 m river stretch more than 100 m river length (i.e. >4 segments) with PEC > RAC (network analysis);
Up- and downstream water bodies included in network analysis for potential hotspots	d			Static: 2000 m water bodies up-and down-stream from the last segment with PEC > 0

a. Degree of protectivity (conservativity) of parameter estimation: percentile of value (in case that an exact determination of a percentile is possible or defined by the methodology of derivation)
b. Protectivity “yes” means: the value(s) are chosen beyond the mean or median of its distribution, but their degree of probability (an xy-percentile) cannot be identified exactly.
c. Geo-referenced variable: **y**es – **n**o
d. Please note: The w/d ratio of 3.3 : 1 is used only for the technical GIS implementation of the HotSpot identification (task of chapter 5) as an preliminary value. For further geo-referenced PRA a mean ratio of 6.6 : 1 for static ditches in Germany is proposed (see chapter 4.4.1 and Table 4-23).

5.3.1 Generation of domain data

At the moment the GeoRisk spatial databases exclusively[3] consist of information derived from the analysis of the authoritative spatial information system ATKIS Basis-DLM (BDLM)[4] (AdV 2003).

Arnold (2010) gives the following condensed description of this information system. The BDLM is the basic digital landscape model with a scale between 1:10.000 and 1:25.000. The responsibility for the continuous update lies with the Regional Survey Authorities (Landesvermessungsämter) who continuously deliver the data to the BKG. The data model of the BDLM consists of point, line and polygon feature types and is thematically categorised into layers, such as built-up areas (e.g. SIE02_F, SIE03_F), vegetation (e.g. VEG01_F, VEG02_F) etc. These layers again contain various feature types (e.g. 2112 industrial and commercial sites, 4102 grassland) which are differentiated by attributes. In some cases – depending on the given situation – the data model allows geometry overlapping of multiple layers, which means that one single landscape element is mapped by two or more feature types out of two or more ATKIS layers. For some situations the ATKIS feature type catalogue[5] (AdV 2003) even demands the overlapping of covering layers over ground layers. The minimum mapping unit of the BDLM depends on the feature type and ranges from 0.1 ha to 1 ha. The main source of information is aerial photography with a resolution of 20x20 cm or 40x40 cm. The interpretation process is supported by ground truthing through the land Survey Authorities, i.e. the interpretation results are compared to the real situation at the corresponding point of time.

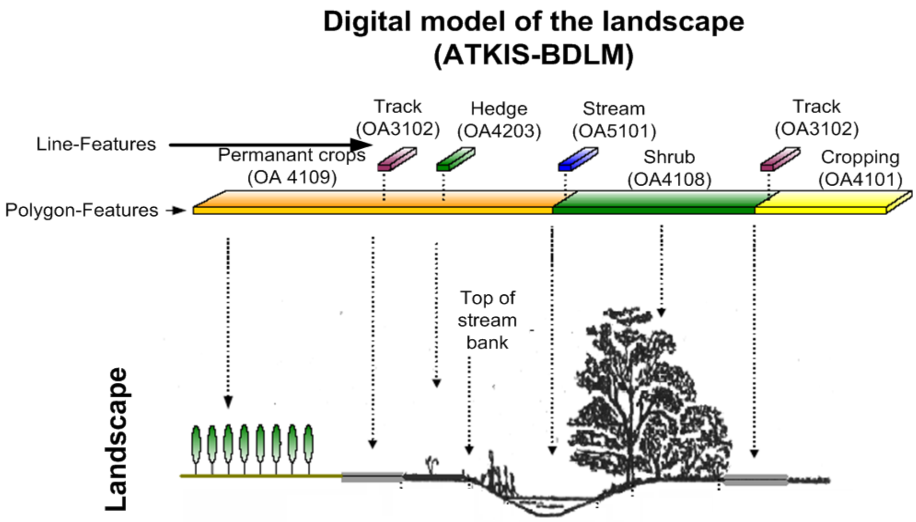


Figure 5-1: Representation of the landscape in the data model of the BDLM

[3] Nevertheless the GeoRisk data model is designed to permit successive integration of refined information in order to describe the exposure situation most realistic. Also concepts for additional data integration have been developed and are subject of chapter 5.5

[4] Spatial data of the geoPERA project (Dechet et al. 2007) were not accessible for the GeoRIisk project.

[5] [germ.] ATKIS-Objektartenkatalog

The creation of the GeoRisk databases follows a general workflow that is shown in Figure 5-2. For each type of permanent crop (fruit, vine, hops) a separate database is built up. In the technical documentation (see Appendix) the general workflow presented here is broken down to the single steps of the necessary spatial data management processes. There are numerous commercial and open source software products that can be used for performing these tasks[6]. In this project the functionalities of Oracle Spatial 11g where used.

Figure 5-2: General workflow for creating the GeoRisk databases

5.3.2 Necessary pre-processing

The workflow starts with the BDLM data delivery, storing in the database and data validation and pre-processing. These steps are taken for the entire BDLM data set. Besides the known but inherent obstacles of the BDLM data model (Röber et al. 2009, Wegehenkel et al. 2006) the validation process revealed redundant features which was met in the pre-processing (see Appendix Technical Documentation). Missing data was discovered in a number of tiles for the BDLM water layer (GEW01_L). This was solved with an additional data delivery by BKG.

5.3.3 The GeoRisk Network Databases

The GeoRisk Network Databases (NDB) store topological surface water networks of the water bodies that are considered in the GeoRisk approach. The core objects are water bodies in a distance of = <150 m to application sites. For network analysis of hotspots (see chapter 6) 2000 m up and down stream of the core objects are included. NDB are built for each of the crop types separately (Figure 5-3). They consist of the BDLM object types for streams (AOA 5101) and ditches (AOA 5103). Both types are included with their line and polygon features (GEW01_L, GEW01_F). Lakes and ponds (AOA 5112) are represented by their surrounding representing the riparian zone which is subject to the GeoRisk exposure and risk assessment. The water body segments have a general[7] length of 25 m.

5.3.4 The GeoRisk Landscape Databases

The GeoRisk Landscape Databases (LDB) store different information necessary to perform the drift deposition simulation. For all core objects[8] the distance to the nearest application site and

[6] Steininger and Bocher (2009) give a detailed overview on existing free and open source desktop GIS projects. The GIS-Report (Harzer 2009) focuses on the description of commercial sofTWAre solutions.

[7] Minor variations occur at the end of a line feature as the total length is generally not a multiple of 25 m.

[8] water bodies in a distance of =<150 m to application sites

each main wind direction (0°, 45°, 90°…315°) up to a distance of 150 m were determined. The distance to the nearest application site was stored for the buffer zones of 3 m, 5 m and 10 m separately. For all core objects and wind directions the information on the presence of filtering vegetation was also stored.

5.3.5 The GeoRisk Exposure Databases

The GeoRisk Exposure Database (EDB) stores the 90th percentile of the GeoRisk drift deposition simulation as the parameter deposition rate indicator for the buffer zones 3 m, 5 m, 10 m.

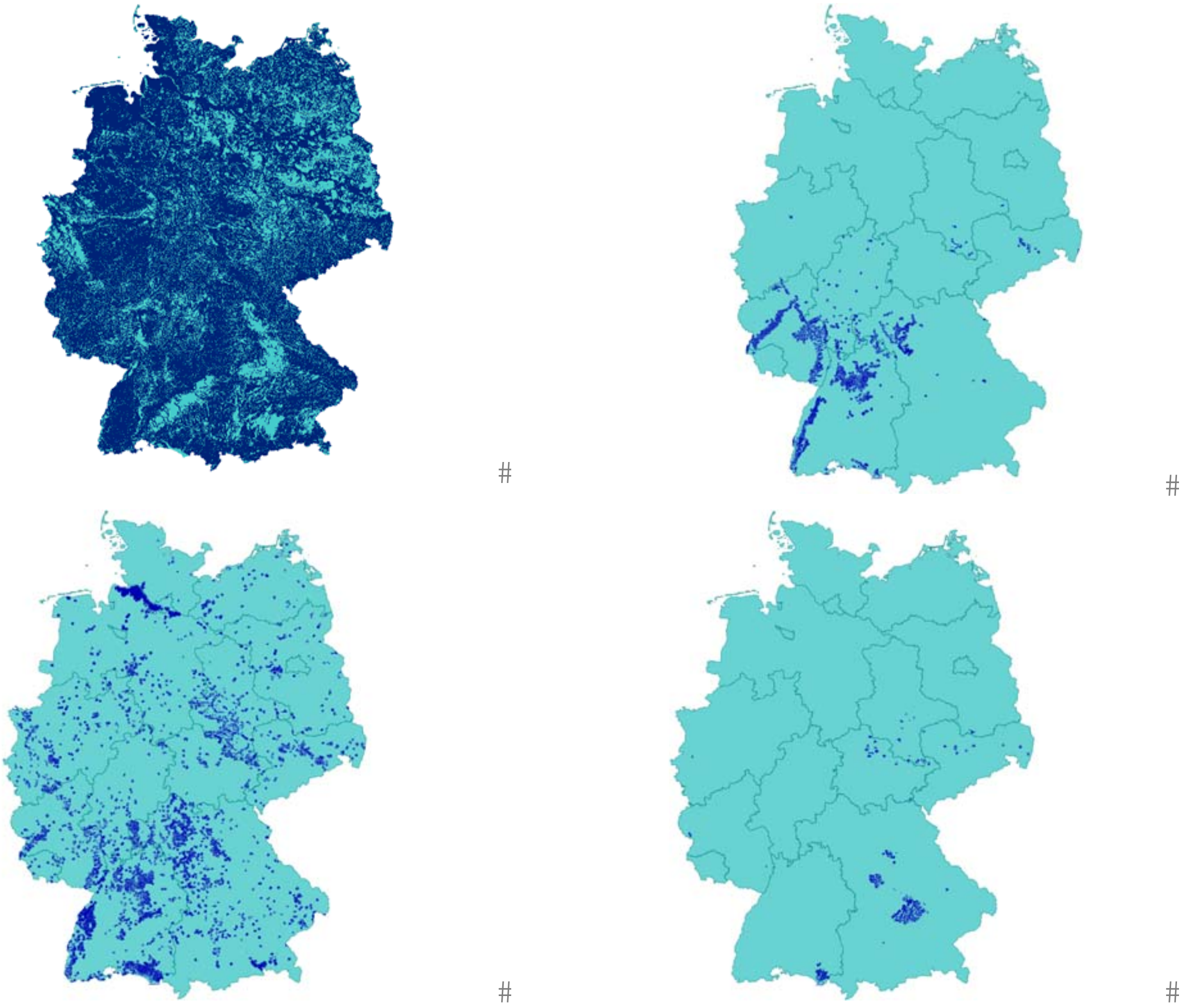


Figure 5-3: Surface water network involved in GeoRisk analysis. UL: BDLM surface water network (AOA 5101, 5103, 5112). UR: GeoRisk NDB for vine. LL: GeoRisk NDB for fruit. LR: GeoRisk NDB for hops.

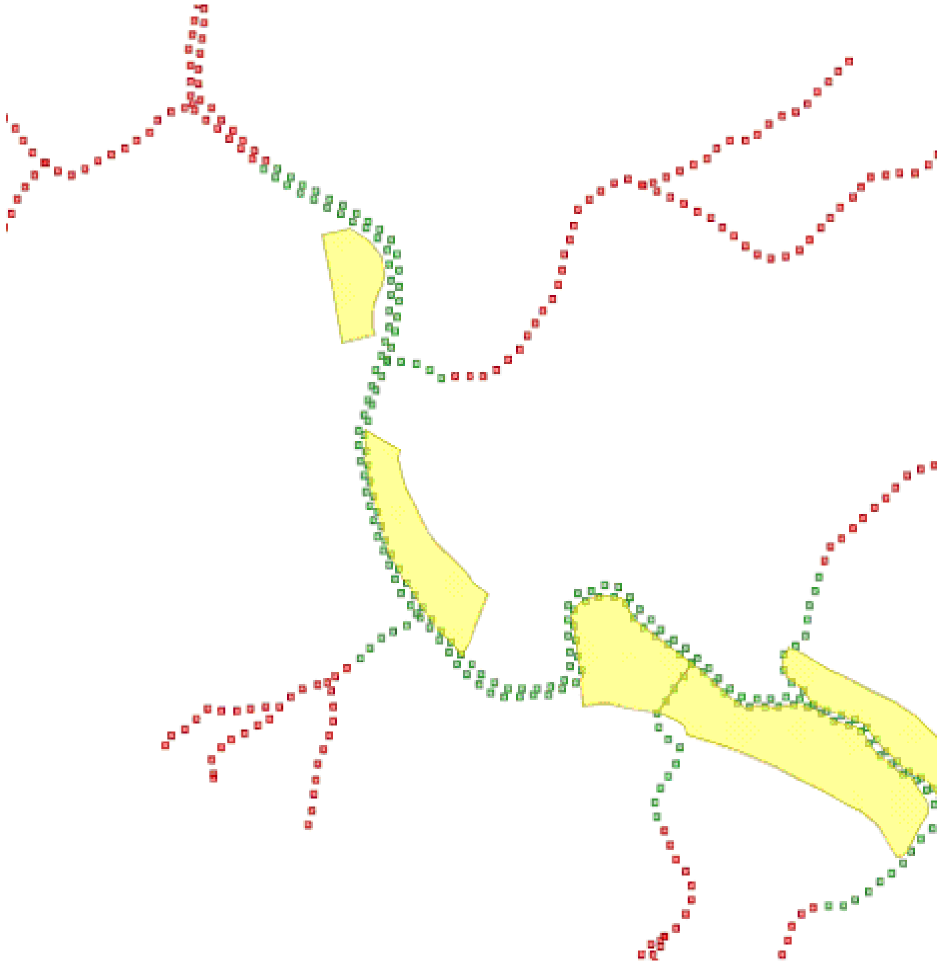


Figure 5-4: Cut-out representation of GeoRisk NDB. Green: segments =<150 m distance to an application site of the target crop (yellow – not content of the NDB). Red: segments >150 m and <2000 m distance to an application site.

5.4 Characterisation of uncertainty and errors

Uncertainty refers to a *lack of knowledge* about specific factors, parameters, or models (U.S.EPA 1997). In the following chapters model uncertainty due to the necessary simplification of real-world processes or phenomena is analyzed:

- Uncertainties in BDLM data on the presence of hedgerows
- Uncertainties in BDLM data on the presence of surface waters
- Uncertainties in BDLM data on the presence of application sites
- Uncertainties in receiving water volume for stagnant ditches
- Uncertainties in the drift deposition indicator
- Uncertainties in the distance calculation

In all cases the approaches for estimating the uncertainties are carried out for sample sites or regions.

5.4.1 Uncertainties in BDLM data on the presence of hedgerows

For assessing the uncertainty in the BDLM data on the presence of hedgerows (OA 4202) the BDLM data and the results of on-screen true colour aerial photo interpretation at the orchard region Lake Constance are compared. The use of metadata[9] assured that the temporal accuracy of both data sets allowed a comparison. The data sets stem from the years 2006 and 2007 (Figure 5-5).

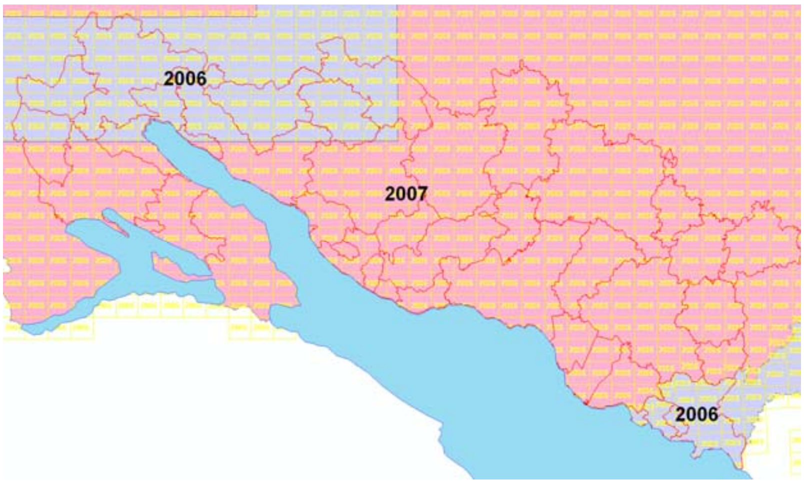


Figure 5-5: Temporal accuracy of BDLM data (black) and DOP (yellow) at Lake Constance

The photo interpretation is supported by a software tool[10] that shows the relevant water body segment in the center of the image at a fix scale (Figure 5-22). In total 5337 water body segments were analysed. For small streams or ditches it is not always visible from an aerial photo perspective whether a hedge is present on one or both sides of a water body. For this reason a hedgerow was considered to be present if it was detectable on either one or both sides of the water body. For comparability reasons the same procedure was applied to the BDLM data, although the dataset explicitly distinguishes between the presence of a hedgerow at either side of a water body (Figure 5-6).

[9] Metadata on the temporal accuracy of both datasets were provided by BKG

[10] developed at JKI

Figure 5-6: Representation of hedges (OA4202) in the BDLM (green dotted line)

The results show that shielded vegetation is not represented in the BDLM for 24 % of the water bodies (Figure 5-7), whereas all hedgerows represented in the BDLM were also recognized in the aerial photo interpretation. This outcome shows that hedgerows are underrepresented in the BDLM which is obviously due to the minimum mapping unit (MMU)[11]. For the exposure and risk assessment of GeoRisk this means that incorporating the parameter filtering vegetation based on hedges of the BDLM is protective. But in real landscapes more water bodies are shielded by hedges than the BDLM would suggest. This supports the findings of geoPERA project (Trapp & Thomas 2009), who compared the results of an object-oriented image analysis with the BDLM data. On the other hand incorporating remote sensing information might lead to misinterpretation when it is necessary to decide which side of the water body is shielded. For this reason, field checks are an important component in the verification of potential management segments.

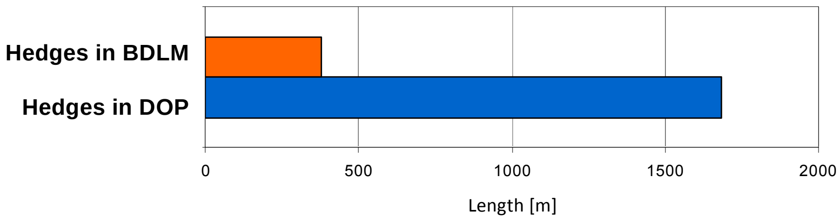


Figure 5-7: Quantitative comparison of the BDLM data and the results of aerial photo interpretation on the presence of hedgerows.

[11] According to AdV (2003) hedges are completely represented from a length of >= 200 m on, or if they characterise the landscape.

5.4.2 Uncertainties in the BDLM data on the presence of surface waters

The spatially explicit GeoRisk approach assesses the risk of water bodies that are present in the GeoRisk databases. At the time of the project GeoRisk databases consist of the following BDLM features: Stream, River, Brook (OA5101); Ditch, Channel OA5103 Lake, Reservoir, Pond (OA5112).

There are two reasons why surface waters might not be mapped in the BDLM:

1) The length or area of the water body is below the MMU. The MMU depend on the attribute value "HYD" (Table 5-2).
2) The water body is missing although the MMU is met. In these cases it is considered as an error of the data set.

Table 5-2: Object definition and minimum mapping unit of BDLM features (AdV 2002).

Object feature	Attribute value	Minimum Mapping Unit (MMU)
Stream, River, Brook **OA5101**	Permanently containing water (HYD 1000)	completely
	Non-permanently containing water (HYD 2000)	Length >= 500 m*
Ditch, Channel **OA5103**	Permanently containing water (HYD 1000)	Completely
	Non-permanently containing water (HYD 2000)	Length >= 500 m*
Lake, Reservoir, Pond **OA5112**	Permanently containing water (HYD 1000)	Area >= 0,1 ha*
	Non-permanently containing water (HYD 2000)	Area >= 0,1 ha*

* According to Golla et al. (2002) some Federal States map objects below MMU stated in AdV (2002).

In order to estimate the importance of this shortcoming a solely data based approach was conducted. A field cross-check of the interpretation results was not performed. Randomly selected survey sites were interpreted manually using on-screen interpretation of true colour orthophotos (fruit n = 199; hops n= 98 vine n = 1710). A survey site was defined by the application site itself and a buffer of 150 m around it [12] (Figure 5-9). Technical stuff with expertise in photo interpretation analysed the sites for potential water bodies or indications of potential water bodies (e.g. linear strips of shrub). All possible or identified water bodies were digitised (Figure 5-9). The working scale was 1:1000 m. In total 39 620 ha agricultural landscape were interpreted. For the different permanent crop types the survey area represents between 1 % and 12 % of the total theoretical drift area in Germany (Table 5-3).

[12] For comparison: A crop site of 10 ha results into a survey site of about 35 ha.

Table 5-3: Statistic characteristics on the survey sites per crop type

Crop type	Count	Area min. [ha]	Area max. [ha]	Mean [ha]	Area sum [ha]	Fraction of total buffer area* [%]
Vine	1710	7.2	123.4	19.4	33 225.4	11.8
Hops	98	10.7	77	24.4	2 387.9	3.1
Fruit	199	9.1	53.8	21.8	4 007.3	1.1

* Indicates the fraction of the total non-overlapping buffer area (150m) around all permanent crop sites of the respective crop type.

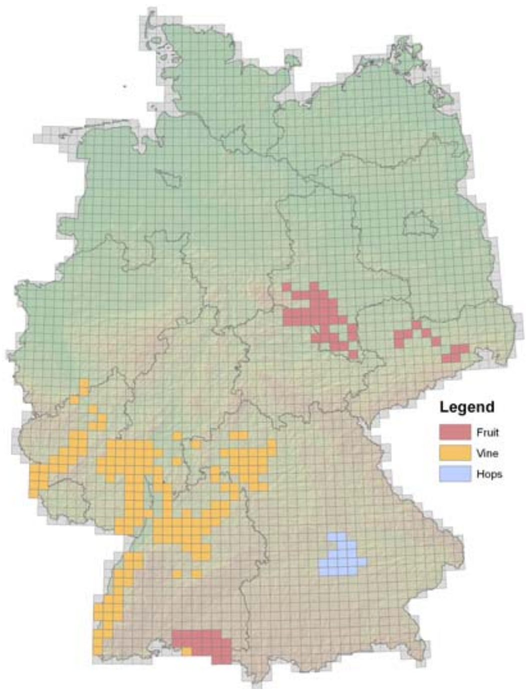


Figure 5-8: Coverage of the survey sites per crop type based on topographic map TK25 tiles

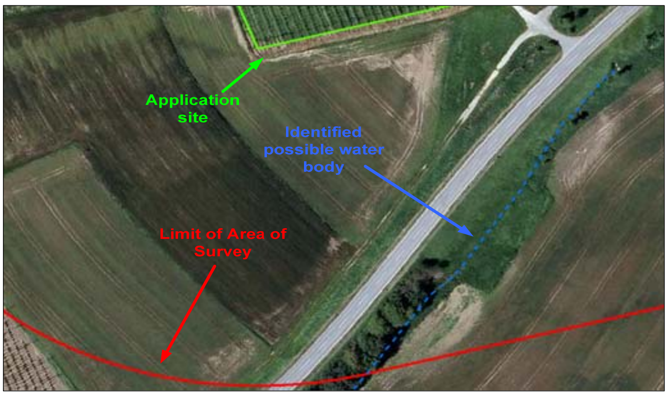


Figure 5-9: Representation of the interface for the aerial photo interpretation

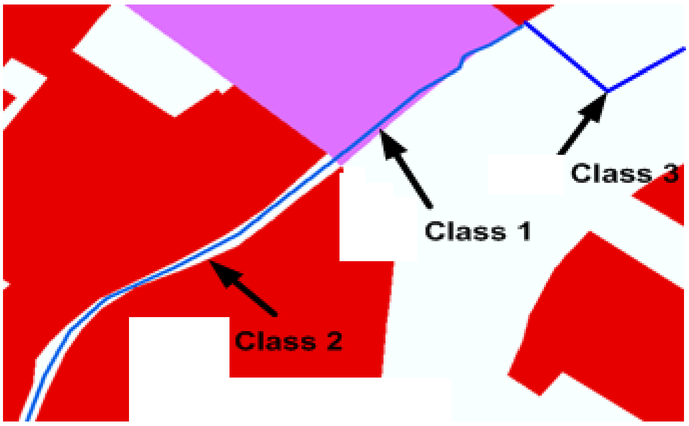


Figure 5-10: Distance classes indicating the probability of a water body to be exposed due to distance information according to BDLM

In a second step the additional water bodies were classified according to the probability of being exposed to spray drift. For this step only distance information was taken into account. Shielding vegetation was not considered. Spatial overlay and distance functions were conducted on the interpretation results and BDLM data. Three distance classes are defined (Figure 5-11):

Distance Class 1: Water bodies that overlay with BDLM application sites indicate a possible high risk.

Distance Class 2: Water bodies lying in a corridor between application sites indicate a medium risk.

Distance Class 3: Water bodies outside a 10 m buffer from the application sites indicate a low risk.

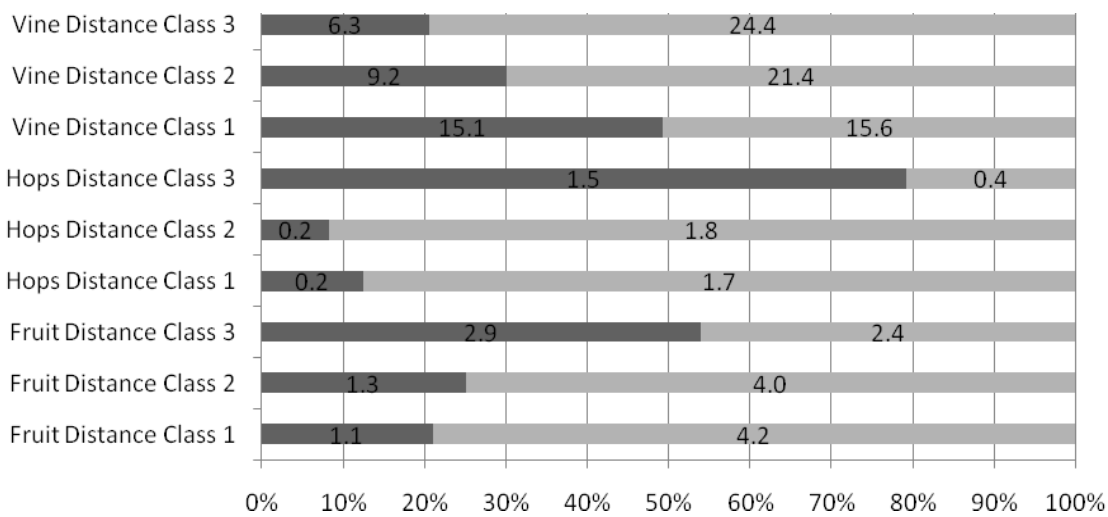


Figure 5-11: Results of the aerial photo interpretation: Additional water bodies (dark grey) or indications for water bodies (light grey) classified according to the distance to application sites. Numbers in the pillars indicate the length in [km].

Table 5-4: Statistic characteristics of potential water bodies within the survey sites

Crop type	Total length* in 150 m buffer [km]	Total length in 10 m buffer* [km]	Distance Class 1 [km]	Distance Class 2 [km]	Distance Class 3 [km]
Vine	291	103.1	15.1	9.2	6.3
Hops	25.4	5.1	0.2	0.2	1.5
Fruit	69.6	14.6	1.1	1.3	2.9

* according to the BDLM

For the interpretation of the results it is assumed that the BDLM data together with the interpretation results represent the total of water bodies present at a survey site. To assure this assumption field checks need to be carried out. The results show that the BDLM covers 89.5 % of the water bodies within the survey sites for vine, 92.5 % for hops and 92.4 % for fruit. Looking only at water bodies within a 10 m buffer to application sites the mapping results are poorer for vine (76.4 %) and fruit (83.6 %), whereas the matching results for hops are about the same (92.2 %). A GeoRisk approach solely based on the BDLM data maps the predominant fraction of water bodies in the landscape.

As the interpretation results are not cross-checked in the field, the outcomes give a conservative estimation of the probability that relevant water bodies are not considered the GeoRisk approach which is based solely on the BDLM data. Before extrapolating the results field checks should be done. For vine a field check is prepared together with ZEPP[13] but not yet conducted. There, also shielding vegetation and hydrological parameters will have to be collected in order to give a comprehensive view on the risk situation of a given water body.

5.4.3 Uncertainties in the BDLM data on the presence of application sites

Application sites that are not represented in the BDLM may still pose a risk to water bodies included in the BDLM. On the other hand application sites according to the BDLM may be abandoned and are not posing a threat anymore.

For estimating the uncertainty of water bodies to be represented in the BDLM but not being considered in the GeoRisk approach, BDLM data on the presence of orchards (AOA 4109, KLT4000) are analysed against data on orchard sites from the agricultural spatial domain system InVeKoS[14]-GIS (BNK-Code “OB”). The InVeKoS is a system that is set up European wide in the context of the financial aid that the European Union grants to producers of certain kind of crop. For these purposes all member states established an Integrated Administration and Control System (IACS). The system consists, among others, of a computerized database and a geographical identification system for agricultural parcels. In Germany the geographic part of the system is referred to as InVeKoS-GIS. The systems are maintained by the authorities of the Federal States (Osterburg et al. 2009, Enzian and Golla 2006). Data policy is heterogeneous.

[13] Central Institution for Decision Support Systems in Crop Protection

[14] Integriertes Verwaltungs- und Kontrollsystem

The analysis is conducted with InVeKoS-GIS data of 2009 of the Federal State Thüringen that are available for download[15]. In a first step a spatial intersection of BDLM application sites with agricultural parcels from InVeKoS-GIS was computed. InVeKoS features without a change in size are considered to be additional application sites which are not represented in the BDLM. In total 640 additional sites were identified. By using a spatial distance calculation a 10 m buffer zone around all additional sites was created. The buffers were then analysed for BDLM water bodies (line features of AOA5101 and 5103). Within this distance there is a high probability of classifying water bodies as risk segments. Additional water bodies were then analysed against already identified risk segments (RS).

Table 5-5: Statistic characteristics of additional InVeKoS-GIS orchards and BDLM water bodies within a 10 m buffer zone

Data set	Count	Mean	Median	P90	Max
InVeKoS Orchards	640	4.3 ha	1 ha	10 ha	169.1 ha
BDLM water bodies	165	106.03 m	34.4 m	329.7 m	1256 m

In the analysis, 165 water bodies with a total length of 17,5 km were identified. 90 % of the water bodies contribute with a length of less than 330 m to a single buffer zone. 50 % contribute with less than 34 m. The analysis of additional water bodies for their potential to create new or contribute to existing RS shows that 46 water bodies will result into a length of 12.9 km additional RS. Another part is connected to already existing RS and increases the length by 1.7 km. For Thüringen as an example the total amount of additionally identified water bodies due to additional application sites increases the length of RS from 18.6 km to 33.2 km.

Although the analysis is based on comparing landscape models without field checks, the example shows that a relevant proportion of application sites are not mapped in the BDLM. These sites lead to a significant increase of RS. Whether the total length of RS increases can only be assumed, as the vice versa situation was not explored: BDLM application sites which are not mapped in InVeKoS-GIS. In any case application sites of InVeKoS-GIS should be considered in the GeoRisk approach which is water body specific. As the Federal States are the owners of INVEKOS data, the availablity and assessability must be discussed

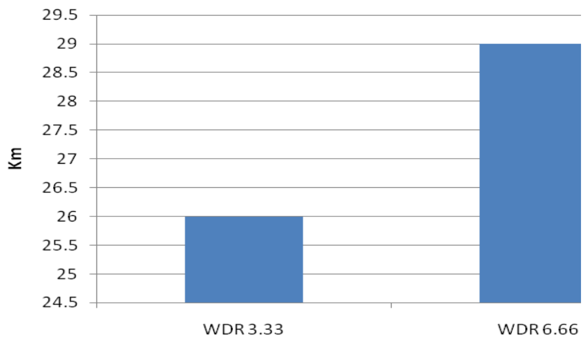
5.4.4 Uncertainties in the estimation of the water volume of stagnant ditches

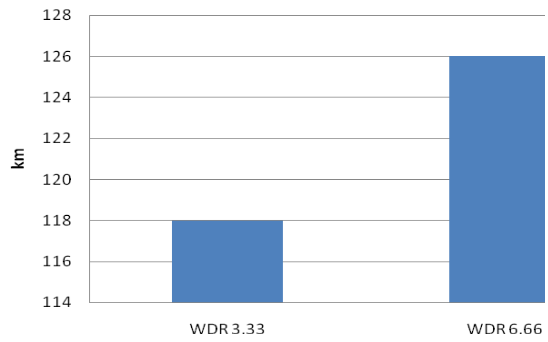
The parameter of receiving water volume is presented and discussed in chapter 4.4. Field data indicate that, besides natural streams, the width to depth ratio (WDR) of stagnant ditches is larger than the estimate of 3.3 : 1 used in current deterministic risk assessment.

In the analysis the influence of a larger WDR on the number of risk segments (RS) for hops and vine is explored; a WDR of 6.6 : 1 is assumed. A database query was conducted on the

[15] http://www.tll.de/mapdown/md_idx.htm

GeoRisk exposure database (EDB) and the exposure concentration of ditches (AOA5103). As the static GeoRisk approach assumes a rectangular cross section profile (see chapter 4.4.3), a general width of 1 m and a WDR of 3.3 : 1, the EDB exposure concentration data can simply be altered within the query by a factor of 2. Only the receiving water volume represented as line features was computed with a modified WDR.





Hops Vine

Figure 5-12: Landscape effect of an increasing width/depth ratio for a stagnant ditch (rectangular profile): Sum [length in km] of RS

The results for hops and vine show that the length of RS increases by 13.5 % for hops and 7 % for vine. Larger WDR effect the length of RS according to static GeoRisk. A future modification of WDR should be regionalized and based on more field data of relevant growing regions.

5.4.5 Uncertainties in drift deposition rate calculation

For estimating the effect of different statistical parameters indicating the drift deposition of an individual water body segment in combination with ways of drift deposition calculation three scenarios were compared using the total length of resulting management segments (MS) as indicator.

(1) **DRIFTmean scenario:** This scenario represents the approach used in the geoPERA project (Schad et al. 2006). A deterministic drift load based on the JKI drift function (JKI 2008) is calculated for each of eight wind directions using spatially explicit distances. The statistical parameter indicating the drift deposition of an individual water body segment is the computed mean of eight drift deposition values.

(2) **GeoRisk scenario:** This scenario represents the simulation approach of GeoRisk for calculating a drift deposition distribution (see chapter 4.2.4). The statistical parameter that indicates the drift deposition of an individual water body segment is the computed 90th percentile of the simulated drift distribution.

(3) **DRIFTmax scenario:** The drift loading in a scenario is calculated according to the DRIFTmean scenario except that the statistical parameter indicating the drift deposition of an individual water body segment is the computed maximum of eight drift deposition values.

All three scenarios have common context parameters: They are computed for an orchard - early stage - scenario and use spatial data of the Lake Constance region from BDLM for distance calculation. A PEC is calculated assuming a rectangular cross section profile, a general width of 1 m and a width to depth ratio (WDR) of 3.3 : 1.

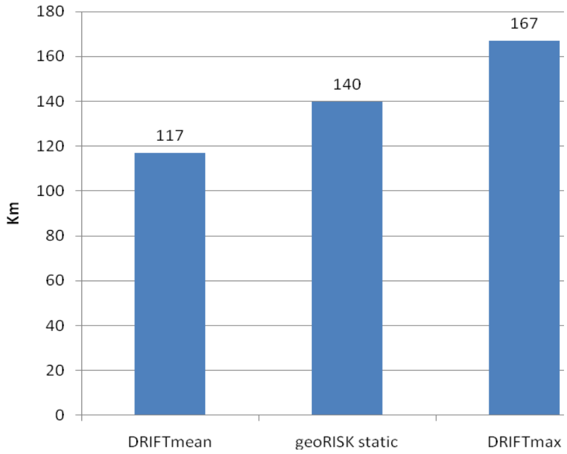


Figure 5-13: Results of different approaches to calculate and indicate drift deposition for an orchard (early stage) scenario: Length [km] of management segment calculated.

In comparison to the length of MS according to the GeoRisk static approach, the $DRIFT_{mean}$ scenario results into 16.4 % less MS. The $DRIFT_{max}$ results into 19.3 % MS more. As the context parameters of the scenarios are the same, it can be assumed that the percentage results can be transferred to other types of permanent crops and to the national results given in chapter 7. However, the results have no consequences for the GeoRisk approach. But they were necessary for discussions in the early stage of the project and to give an estimation of the effects of different statistical parameters that indicates the drift deposition of an individual water body segment and ways of drift deposition calculation.

5.4.6 Uncertainties in the distance calculation

5.4.6.1 Distance calculation within the ATKIS Model

The deposition rate is sensitive for the parameter edge of field – edge of water body distance (see chapter 4.2). Distance information in the GeoRisk approach is currently derived from BDLM data. The ATKIS data model does not contain objects for the transition zones between surface waters and application sites unless the area of this zone is larger then the MMU of the object e.g. grassland (AdV 2002, AdV 2008). Therefore the distance of fields which are directly adjacent to surface waters are assumed to be 3 m in perpendicular direction (for simulated buffer regulation: 5 m and 10 m respectively) (see chapter 5.3.4).

Although the metadata of the BDLM gives an absolute spatial position error for line features of +/- 3 m, it is not feasible to use it for distance calculation. This is because the buffer distance is a model assumption and not a location specific phenomenon (e.g. measured from aerial photographs).

5.4.6.2 Distance calculation with spatial data of different data models

Nevertheless a probabilistic simulation of the deposition rate can benefit from incorporating uncertainty in the measurement of distances if the information stem from measurements in the field or from remote sensing images or aerial photographs. During the project two different approaches were examined for incorporating uncertainty in distance calculation with objects that stem from different data models.

I. Geometric uncertainty of spatial objects

For a future integration of additional data that come from outside the ATKIS data model and have no topological relationship with ATKIS objects, metadata on the absolute spatial position error must be provided. The mathematic basis for deriving the distance $dist$ and standard deviation Δf is given with the equations in Appendix of chapter 5.

In order to estimate the effect of incorporating uncertainty due to the variability of distance measurements the GeoRisk model for deposition simulation was extended with the standard deviation Δf of the distance measurements. Table 5-6 compares the results of hypothetic integration scenarios for different edge-of-field distances and different absolute spatial position errors with results of deposition rates calculated inside the ATKIS model with the GeoRisk drift deposition simulation model.

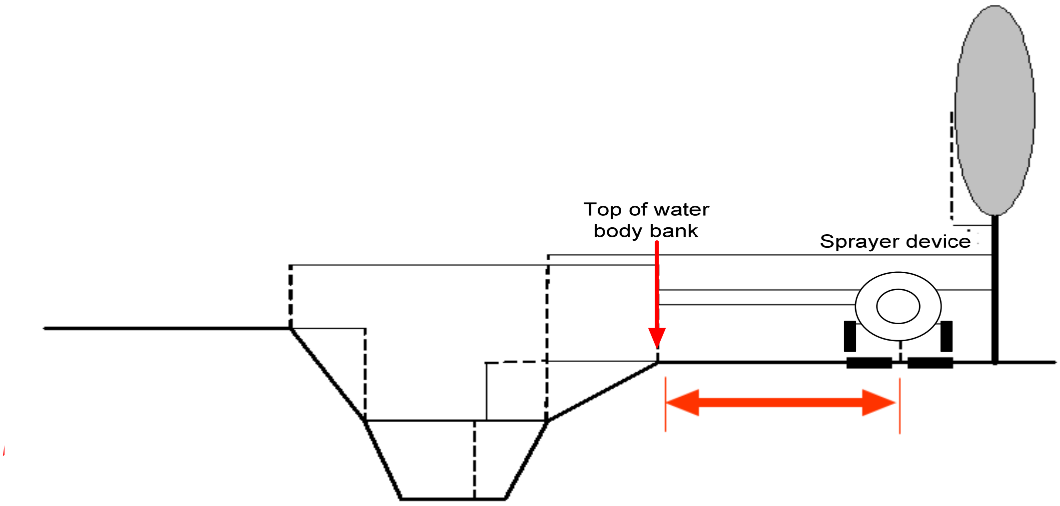


Figure 5-14: Relevant edge-of-field to water body distance for drift deposition calculation in compliance to the drift BBA measurement protocol (1992)

Table 5-6: Results of the GeoRisk deposition simulation for orchards extended with information on the absolute spatial position error of distance measurements

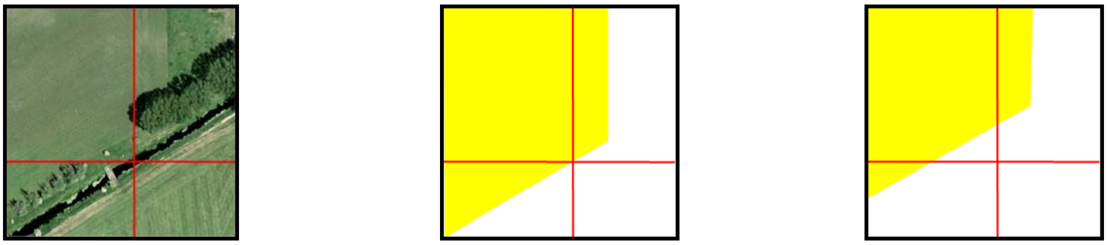
	Data integration scenario	Edge-of-field distance [m]	Std.dev. [m] object A e.g. edge-of-field	Std.dev. [m] object B e.g. stream	Total error [m]	Max. error [m]	90-P deposition rate (total) [%]	90-P deposition rate (max) [%]
1	Within one DLM* e.g. BDLM	3	*n.a.*	*n.a*	*n.a*	*n.a*	30.4	30.4
2	Two different DLM	3	1	1	1.41	2	29.7	>30.4
3	Within one DLM e.g. BDLM	5	*n.a*	*n.a*	*n.a*	*n.a*	23.5	23.5
4	Two different DLM	5	1	1	1.41	2	24.9	30.4
5	Two different DLM	5	1	2.5	2.69	3.5	27.2	>23.5

* Digital landscape model

The calculations show that the integration of additional information on the absolute spatial position error in the deposition model affects the results of drift deposition rate calculations. Only scenario 2 leads to a slightly lower deposition rate (2.3 %) compared to scenario 1. In all other scenarios the deposition rates of the extended models lead to higher deposition rates in the 90th percentile of the distribution: up to 29.4 % compared to scenario 1 and 3. Therefore the GeoRisk deposition model should be extended for incorporating uncertainty in the measurement of distances before integrating data of different landscape models (see chap 5.6).

II. Multicriteria assessment using distributed spatial data sets

A more comprehensive approach was developed within a masters thesis and contributed to the project. The approach allows to integrate multiple additional data sets and combine different spatial and temporal resolutions (Krumpe 2010). Due to the fuzziness of the geospatial quality dimensions, fuzzy sets are applied and a multi-criteria assessment is conducted. As a result a harmonized (integrated) spatial data set is created which is again the basis for a landscape analysis and GeoRisk deposition rate simulation. The data integration part is based on established methods of fuzzy membership functions and fuzzy rule systems to incorporate geometric, thematic and temporal aspects of uncertainty.



Aerial photo of a water body in distance to an application site. The cross indicates the axis of the water body.

Representation in the ATKIS model: water body directly adjacent to application site. A minimum perpendicular distance of 1 m is assumed.

Representation of the result of the multicriteria assessment using different additional data sets: A perpendicular distance of 11 m is calculated.

Figure 5-15: Concept of a multicriteria assessment using distributed spatial data sets

For 30 randomly selected water body segments near orchards with an in-between distance of at least 500 m the surrounding landscape structure is refined by automated integration of different[16] web feature service (WFS). Based on the derived information exposure concentration distributions are calculated according to the GeoRisk static model. The results are compared with exposure concentration distributions based solely on the BDLM data (Table 5-7).

In most cases exposure simulations based on the harmonised data lead to lower exposure concentrations. This is due to a better differentiation of the distance between application sites and water body but also better representation of shielding vegetation obtained from the web feature services (WFS) “Landschaftselemente”. In cases where an application site is not mapped in the BDLM but in the InVeKoS-GIS data set a higher exposure concentration is calculated.

The approach proved to be applicable and can serve the UBA as a method for maintaining the GeoRisk landscape database according to known web map and web feature services.

[16] ALK, InVeKoS-GIS “Feldblock”, InVeKoS GIS “Landschaftselemente”

Table 5-7: GeoRISK static exposure simulation (orchard, day 90) for BDLM and a harmonized data set based on a multicriteria assessment

ID	GeoRisk PEC static based on BDLM [µg/l]	GeoRisk PEC static based harmo-nized data set [µg/l]	Delta [µg/l]		ID	GeoRisk PEC static based on BDLM [µg/l]	GeoRisk PEC static based harmo-nized data set [µg/l]	Delta [µg/l]
1	91.71	64.92	26.79		16	26.57	00.81	25.76
2	74.59	20.59	54.00		17	73.20	06.37	66.83
3	73.47	51.86	21.61		18	75.09	20.89	54.20
4	74.48	73.46	01.02		19	73.08	17.82	55.26
5	04.25	01.20	03.05		20	75.10	29.78	45.32
6	73.45	02.79	70.66		21	73.97	76.21	-02.24
7	75.80	74.22	01.58		22	09.94	01.80	08.14
8	37.03	01.14	35.89		23	74.69	30.16	44.53
9	73.95	36.99	36.96		24	21.81	07.67	14.14
10	73.13	36.97	46.16		25	17.33	00.47	16.86
11	00.00	02.34	-02.34		26	91.75	57.82	33.93
12	00.00	18.35	-18.35		27	95.15	95.00	00.15
13	73.56	22.43	51.13		28	74.18	11.45	62.73
14	26.57	00.81	25.76		29	74.24	50.02	24.22
15	73.20	06.37	66.83		30	00.00	74.23	-74.23

5.5 Technical concepts of GeoRisk

5.5.1 Concept of the domain system GeoRisk-WEB

The GeoRisk approach is implemented as a web-based application, GeoRisk-WEB. The governmental IT coordination division[17] advises in several initiatives (e.g. E-Government 2.0[18], Deutschland-Online[19], BundOnline 2005[20]) the consideration of web-based applications as they are in principle useable on any operating system and useable without additional software and license requirements. As long as the software engineering considers technical standards, innovative applications can be integrated in existing administration processes.

The application allows the classification of pesticide in risk mitigation groups (RMG) according to their ecotoxicity for aquatic organisms. The assessment bases on the parameters date of the first application, amount of active ingredient and RAC. The result of the calculation states for all RMG at once whether and to which extent[21] MS occur. The results can either be downloaded as report (pdf-format) or as an open spatial data file (GML[22]). Besides these functions GeoRisk-WEB integrates a tool to apply the GeoRisk model for the calculation of drift deposition and exposure concentration in a static water body.

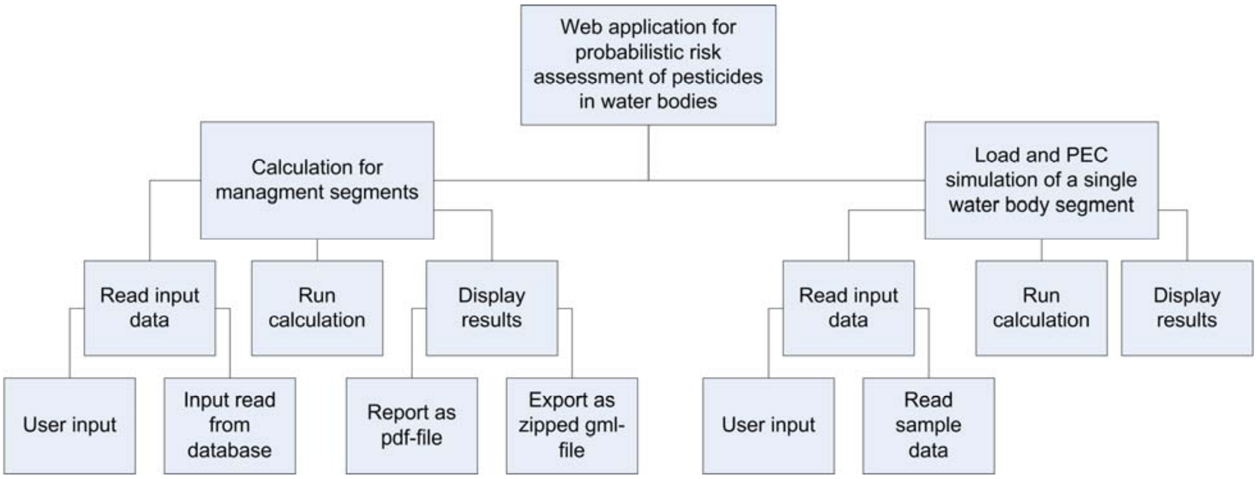


Figure 5-16: Functional hierarchy diagram of GeoRisk-WEB

[17] „Die Beauftragte der Bundesregierung für Informationstechnik“ http://www.cio.bund.de/cln_102/DE/Home/home_node.html [access 13.4.2010]

[18] http://www.cio.bund.de/DE/E-Government/E-Government-Programm/e-government-programm_node.html

[19] http://www.standardisierung.deutschland-online.de/Standardisierung_Internet/broker

[20] http://www.cio.bund.de/cln_102/SharedDocs/Publikationen/DE/E-Government/abschlussbericht_bundonline_2005_download.html

[21] Given is the length kilometers

[22] The Geography Markup Language (GML) is the XML grammar defined by the Open Geospatial Consortium (OGC) to express geographical features (http://en.wikipedia.org/wiki/Geography_Markup_Language)

5.5.2 System architecture

Due to the high spatial and temporal resolution (e.g. 25 m water body segments, trapeze function defining a weekly drift filtering rate of vegetation, network analysis capacity) the databases of GeoRisk are very large. For performance reasons, the calculation of hotspots and most of the application logic is realised on the DBMS level. The architecture of GeoRisk-WEB is therefore implemented as a two-tier-architecture[23]. But still the system engineering incorporates Model-View-Controller architecture which isolates the application logic from input and presentation.

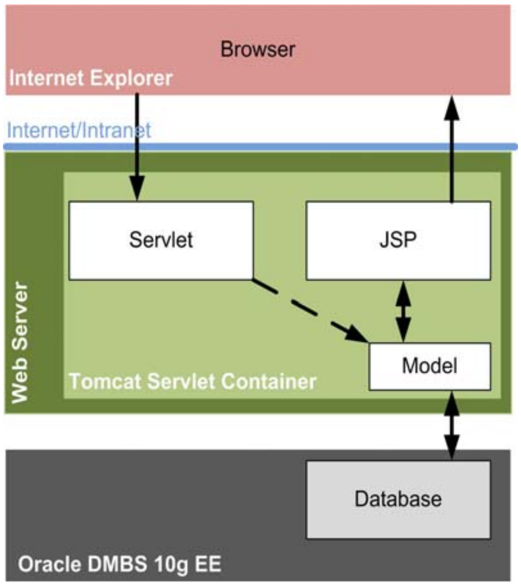


Figure 5-17: Two-tier architecture of GeoRisk-Web

5.5.3 Database Management System

The technical concept of GeoRisk comprises a central database with Oracle spatial as spatial data management system. Oracle supports the ISO TC211 standard which allows the integration of spatial data in distributed systems such as ESRI ArcGIS, ArcView or any FOSSGIS[24] software. Adverse to this, the MS identification is realised with a relational table mapping[25]. The implementation of standards and open software development techniques generally allows the implementation on DBMS different from ORACLE.

[23] For a short introduction see http://en.wikipedia.org/wiki/Multitier_architecture [accessed 26.05.2010]

[24] Free and Open Source SofTWARe Geospatial Information Systems

[25] For reasons of a still open discussion about the DBMS concept at UBA, it was agreed with the UBA to realize MS identification without Oracle spatial algorithms

5.6 Concept of GeoRisk spatial database maintenance

5.6.1 Maintaining the GeoRisk exposure database

The concept of GeoRisk database maintenance includes the processes of periodic updating, backing-up[26] and historisation[27] of data of the GeoRisk databases. The concept is associated with the procedure of teh GeoRisk approach (see chapter 3) and the separation of the MS identification from the legal pesticide registration process. Accordingly, the purposes of the GeoRisk exposure database (EDB) which stores the results of the GeoRisk simulation model are twofold and differ e.g. concerning the updating concept.

First, the EDB serves as basis for the generic[28] identification of management segments (MS). Although this action is not part of the legal registration process, a most complete identification is a precondition for the implementation of GeoRisk in the pesticide registration procedure. This requires that the representation of the presence and location of surface water and sites of permanent cropping is most up-to-date and detailed. Chapter 5.4 focuses on this topic and provides qualitative and quantitative estimations of uncertainties of the ATKIS database.

Another purpose of the EDB is supporting the daily registration work of the registration authorities and the applicants. Using the GeoRisk tool geoWEB the EDB is queried in order to assess the buffer zone restriction for a certain product.

5.6.2 Technical actions in context of generic MS identification

5.6.2.1 Principle considerations

The process of generic MS identification does not affect regulation in so far as registration applicants are concerned. But still the process is of great importance as it will effect in the first instance owners and users of the production sites[29]. Therefore the identification of MS on the one hand must comprise all possible water bodies with a potential unacceptable risk due to spray drift and it must not underestimate the deposition situation e.g. due to a wrong spatial information on hedges. On the other hand the identified MS must be robust in terms of a positive validation in the field. This requires precise information on all spatial parameters for an appointed date of the GeoRisk implementation. **It is very important for the updating process that after implementation it has to be legally assured that changes in the landscape, e.g. by establishing new cropping sites, or due to land reforms[30] etc. will not cause any new water bodies at risk** (see chapter 9). With these preconditions the need for a frequent updating in context of generic MS identification does not exist.

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[26] Because of the technical nature of this topic, refer to the technical documentation, Appendix B of this report, for details

[27] See footnote 11

[28] Ref. to chap. 3 on the concept of generic MS indentification

[29] For more information on stakeholders involves and possible consequences refer to chap. 9

[30] ‘Flurneuordnung’

5.6.2.2 Actions of the registration authorities

The GeoRisk databases built up during the project consist of information that is entirely derived from the ATKIS BDLM. Chapter 5.4 estimates the main shortcomings of the BDLM concerning a realistic spatially explicit deposition calculation at the water segment level. At different other sections of this report other limitations of the ATKIS model in context of GeoRisk dynamic approach are mentioned (see chapter 8).

To support the improvement of the underlying GeoRisk databases two approaches have been developed. The approach of extending GeoRisk with different and distributed spatial datasets was developed in the scope of a Master thesis (Krumpe 2009). It is a comprehensive approach that can be applied for the integration of spatial vector databases from different sources. This approach is described in the following chapter (see chapter 5.6.2.3).

A specific approach for updating the GeoRisk databases is presented in the following mainly from the perspective of the applicant. But registration authorities can use this approach, too. It can, e.g., be a requirement for any research project funded by the regulation authorities that if applicable relevant spatial data retrieved during the project need to be prepared according to the specifications given in chapter 5.6.4.

5.6.2.3 Extending GeoRisk with different spatial datasets

The concepts of integrating different spatial datasets according to GeoRisk requirements[31] is based on a literature[32] study presented in Krumpe (2009). The integration of spatial vector databases of different sources is generally based on the idea of comparing two data sets, while one is used as a reference and a second one is aligned to it (Butenuth et al. 2007). For the integration of multiple data sets it has been shown how corresponding objects can be found when several data sets have to be integrated (Beeri et al., 2005). Due to the complexity of the integration problem it is very difficult to solve this task with one closed system. The traditional way of desktop GIS integration procedure using overlay and distance functions[33] on entire datasets is not appropriate for GeoRisk. as it is difficult to control the spatial quality of the results.

The data integration as part of the developed and proposed approach for GeoRisk is based on established methods of fuzzy membership functions and fuzzy rule systems to incorporate geometric, thematic and temporal aspects of uncertainty (Krumpe 2010). With the increasing availability of data sets methodologies are needed for the efficient integration, evaluation and analysis of serveral spatial data sets on a given spatial local. They have to address the geometric and semantic uncertainty of spatial data (Schiewe 2010, Kinkeldey et al. 2010). A method based on fuzzy roles was developed to challenge these demands (Schiewe 2010) as well as a prototype for practical data harmonization (integration) to test the performance and to evaluate the results of the procedure. For 30 water body segments near orchards the surround-

[31] In this context geoRISK requirements refer to the feasibility (personal expenses) of data integration concepts conducted at UBA

[32] For incorporated literature and internet sources on the topic “spatial data access and integration” refer to Appendix to this chapter.

[33] Described for the spatial and thematic integration of *Gewässerstrukturgütekartierung* in ATKIS BDLM in Golla et al. (2002)

ing landscape structure is refined by automated integration of web feature service (WFS) of ALK, InVeKoS-GIS "Feldblock" and InVeKoS GIS "Landschaftselemente" to ATKIS BDLM (see chapter 5.4.6).

Athough the approach can also be realised with local (undistributed) data[34], the data acquisition process of the proposed approach is based on the concept of spatial data infrastructures. The term "Spatial Data Infrastructure" (SDI) is often used to denote the relevant base collection of technologies, policies and institutional arrangements that facilitate the availability of and access to spatial data. The SDI provides a basis for spatial data discovery, evaluation, and application for users and providers within all levels of government, the commercial sector, the non-profit sector, academia and by citizens in general (Nebert 2004). In Europe, SDI has a legal basis, the so called INSPIRE Directive[35]. The implementation into national law was completed in February 2009 with the law "*Gesetz über den Zugang zu digitalen Geodaten (Geodatenzugangsgesetz)".* Besides the legal requirements for data producers of the public sector for arranging the availability of and access to spatial data according to standards, European initatives in the framework of environmental reporting and monitoring[36] strongly promote this process. As a result a range of web based services is already available through federal authorities (e.g. BW[37], NI[38], RP[39]). At the same time technical and legal aspects of accessing distributed data sources are faciliated. Environmental modelling greatly benefit from this development, although data mostly stem from different sources and scales.

5.6.3 Technical actions in the context of regulation process

5.6.3.1 Workflow of the applicant

It is in the interest of all stakeholders that both information sources, the information of the underlying database for the registration process and the database for the generic MS analysis is state-of-the-art. According to the GeoRisk approach pesticides can only be registered in a specific RRG if there are no relevant[40] water bodies affected in the landscape.

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[34] Need to be published as WFS

[35] Directive 2007/2/EG (INSPIRE);ABl. EU L 108; 25. April 2007

[36] For detailed information refer to internet presentations of the following projects: ESDI, HUMBOLD®, SEIS®, KOPERNIKUS®

[37] http://www.geoportal-bw.de/geodatendienste_lubw_s1.html

[38] http://www.umwelt.niedersachsen.de/index.php?navigation_id=2812&article_id=8871&_psmand=10

[39] http://map1.naturschutz.rlp.de/service_lanis/mod_wms/wms_list.php

[40] For a detailed discussion of the geoRISK approach refer to chap. 3

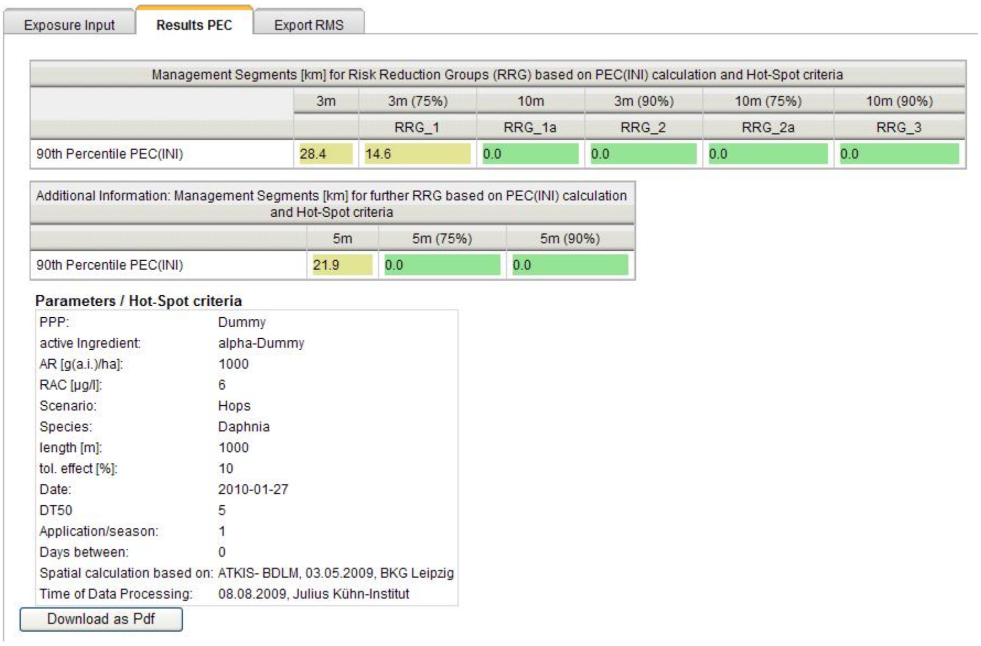

Figure 5-18: GUI of GeoRisk-WEB: Tab for displaying the results of Management Segments for Risk Reduction Groups (RRG)

During the assessment procedure using the web-based application GeoRisk-WEB the applicant receives the information whether and to which extent MS occur. The results are given in length [km] for the different RRG. A product can be registered in all RRG with no MS reported.

If an applicant wants a product to be registered in a lower RRG this can be achieved within a refinement process that is supported within GeoRisk. In this process, the applicant must prove that the exposure situation of MS identified by GeoRisk for specific RRG is different from reality and that these refined data would lead to a lower drift entry.

Three tools are provided to support the workflow illustrated in Figure 5-19:

1) **Spatial data download:** The MS features of a RRG group can be downloaded as open spatial data file (GML[41]). This file provides the applicant with the location of MS for cross-checking.
2) **GeoRisk-Web drift deposition calculator:** The module calculates the drift deposition distributed at the water body segment level according to the GeoRisk specifications. A GUI allows the entry of exposure relevant distances and the presence of drift filtering vegetation (see Appendix, Technical document).
3) **Data specification:** GeoRisk provides data specifications to the applicant in order to provide additional data (see chapter 5.6.4) to the registration authority to be considered in the following update (see chapter 5.6.3.4).

[41] The Geography Markup Language (GML) is the XML grammar defined by the Open Geospatial Consortium (OGC) to express geographical features (http://en.wikipedia.org/wiki/Geography_Markup_Language)

Figure 5-19: Workflow of the applicants' process of receiving spatial data und providing spatial data in the framework of pesticide registration

5.6.3.2 Workflow of the registration authority

For the UBA an efficient updating workflow is of importance. The necessary steps for this are shown in Figure 5-20. One supporting element is the specification of the requirements for additional data (see chapter 5.6.4). The UBA receives the data according to these specifications from the applicant. It is up to the UBA whether to define as well a specific way and media for transmitting the information e.g. via e-mail. The following step of data verification firstly includes a check for compliance with the data specifications. This can be done manually or realized as automated database procedure. The choice for one of the two ways will mainly depend on the quantity of refinement requests which cannot be foreseen today. Additionally, a cross-check on the plausibility of distance and drift filtering vegetation information is to be performed. This will be a manual procedure which can be outsourced. Depending on the quantity of refinement requests it is performed on every request or based on a statistical point selection procedure.

Figure 5-20: Workflow of the UBA process of integrating additional data for updating GeoRisk EDB in the framework of pesticide registration

Verified additional data is stored with all fields in a separate table of the GeoRisk database. It is advised that the UBA defines additional fields to describe the refinement requests (e.g. applicant, date). Apart from the DBMS inherent historization functions this table supports the documentation of the successive LDB update. On DBMS side the update processes and computations can be realized with triggers which automatically execute a predefined event, e.g. a new data entry in a table takes place. The information for a water body segment in the LDB is updated when a data entry with a unique water body segment ID occurs in the additional data table. The update of the drift deposition indicator (90^{th} percentile) for a water body segment is again achieved with first a trigger that executes the deposition simulation according to the GeoRisk specifications. Finally a trigger updates the drift deposition indicator of a water body segment in the EDB. The GeoRisk-WEB application retrieves now updated information for the registration requests.

5.6.3.3 Timeframe of the updating process

A six monthly cycle is advised as updating period as most of the steps of the updating process can be automated (see chapter above). The short time spans between the updates of the GeoRisk EDB will not cause a delay of the registration process. An applicant who provides additional data according to chapter 5.6.4 can apply for a registration based on the provided additional data at the latest after six month. The registration authorities can contribute to the updating process following the approaches of chapter 5.6.1.

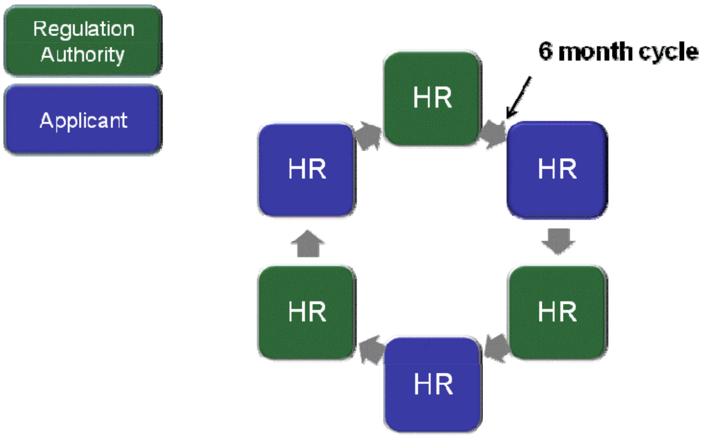

Figure 5-21: Conceptual workflow of the updating process in the framework of pesticide registration

5.6.4 Specifications and requirements on additional data

For an efficient, consistent and provable updating and refinement process additional data need to fulfill certain requirements to be considered. The following list comprises the core requirements.

Unique ID: The unique ID for water body segments in the GeoRisk DB is provided from the GeoRisk-Web system when downloading the spatial data file of the MS. This field functions as a foreign key between an additional data table and the GeoRisk DB.

Distance information: A distance value in [m] has to be provided for each of the eight wind directions to the next application field up to a distance of 150 m. If there is no application site in a direction, this is indicated by the numeric value -1.

Filtering vegetation: A Boolean value (true/false) indicating the presence of drift filtering vegetation has to be provided for each of the eight wind directions.

In order to validate the additional data sets certain metadata need also to be provided.

Method of the landscape analysis: A description of the method of the landscape analysis needs to be provided. The description gives information on the processing for distance calculation and vegetation identification. This includes information on underlying data, systems used, period of processing and the method of the ground truthing.

Relative accuracy: In order to include the uncertainty in distance calculation (see chapter 5.4) in the drift deposition simulation, either statistical parameters of the ground truthing for distance measurements or the individual results need to be provided.

Institution: The institution and a contact person responsible for conducting the landscape analysis needs to be mentioned and, if applicable, the holding of relevant ISO certificates.

5.6.5 Estimation of costs

5.6.5.1 Personnel expenses for running and maintaining the GeoRisk System

It is intended that the GeoRisk database and application will be set up at the UBA. For the estimation of costs the experiences made throughout the project are considered. The cost estimation is divided into two main tasks: *Running the GeoRisk system* and *maintenance of the GeoRisk system*. Running the system in this context covers all actions to keep the database and the web server running or to restart the system. It is provided that backup and the survey of the system is done with automatic routines. The GIS division at the UBA or with contracting external consultants can realize the maintenance of the system. In this context the term “maintenance” covers all actions for updating the GeoRisk database including the extension of the GeoRisk-WEB application validation via configuration files of additional data provided by applicants.

The personnel expenses are estimated as follows:

Table 5-8: Personnel expenses for running and maintaining the GeoRisk system

Position	**Hours per Year**
Database System Administrator	40
GIS-Technician	80 (ATKIS update)
GIS-Technician	8 (HR update)

5.6.5.2 Expenses for MS verification

The estimation of the costs for MS verification is based on an on-screen interpretation of aerial photos. The true color photos stem from the BKG web map service and have a ground resolution of 40 cm. The assessment was conducted for all 141 km of MS in hops. The task of the image analyst is to assess on the water body segment level whether or not a water body can be verified on the images and whether a field is nearby in a distance of up to 15 m. The presence of filtering vegetation was not part of the verification process since relevant details were not clearly identifiable from the image perspective, e.g. whether the vegetation was present on one side of a water body or on both sides or whether the vegetation was without gaps at the relevant height of about 1 m – 2 m.

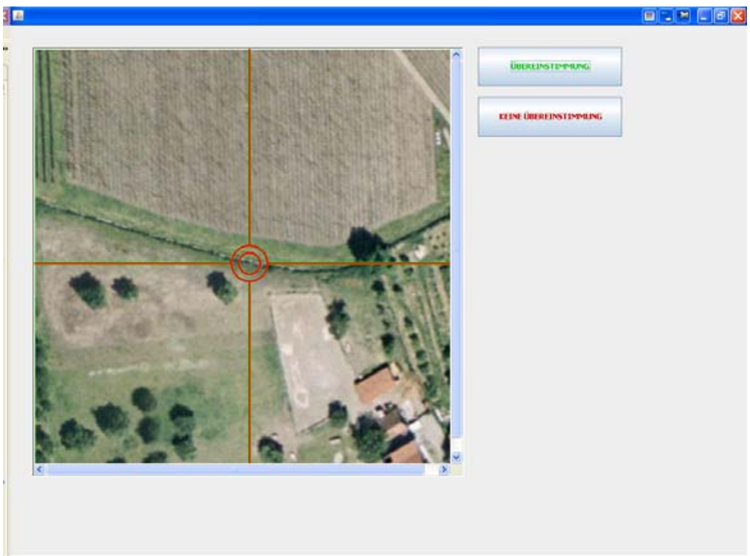

Figure 5-22: GUI of the software that supports MS verification

The work of verification is supported by software [42] that shows the relevant water body segment in the center of the image and two circles indicating the distances of 10 m and 15 m at a fix scale. The analyst affirms or rejects an MS based on the distance information.

For hops 34 % of the total MS length of 141 km is not approved. The costs in personnel expense can be estimated with a total of 0,5 days for a GIS-Technician who is responsible for data preparation and 3 days for an image analyst.

By including filtering vegetation in the analysis according to chapter 5.4.1 additional reduction can be expected without more time expenses.

Table 5-9: Personnel expenses for MS verification of hops

Task	Position	Days
Preparation	GIS-Technician	0,5
Interpretation	Image Analyst	3

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[42] Developed at JKI

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5.8 Appendix

I. Geometric uncertainty of spatial objects

The mathematic basis for deriving the distance $dist$ and the corresponding standard deviation $\Delta dist$ is given by the following common equations:

$$f = f(x_{1,} x_{2,} ..., x_n) \Rightarrow \ function \ of \ measurement \ \ x_1 \pm \Delta x_1, ..., x_n \pm \Delta x_n$$

The error Δf of the function f is calculated as follow:

$$\Delta f = \frac{\partial f}{\partial x_n} \Delta x_n$$

Assume A is the top of the water body bank and B is the relevant reference point in the field according to the drift measurement protocol (BBA 1994), the so-called edge-of-the-field.

Assuming that $\Delta x_B = \Delta y_B \ \ und \ \ \Delta x_A = \Delta y_A$ and that there is no correlation between the measurements of A and distance $dist$, total error $\Delta dist$ and the absolute maximal error $\Delta dist_{max}$ are calculated with the following equations.

$$dist = f(x_A, y_{A,} x_B, y_B)$$

$$dist = \sqrt{(x_B - x_A)^2 (y_B - y_A)^2}$$

$$\Delta dist = \sqrt{(\frac{\partial dist}{\partial x_B})^2 \Delta x_B{}^2 + (\frac{\partial dist}{\partial x_A})^2 \Delta x_A{}^2 + (\frac{\partial dist}{\partial y_B})^2 \Delta y_B{}^2 + (\frac{\partial dist}{\partial y_A})^2 \Delta y_A{}^2}$$

$$\Delta dist_{max} = \left| \frac{\partial dist}{\partial x_B} \Delta x_B \right| + \left| \frac{\partial dist}{\partial x_A} \Delta x_A \right| + \left| \frac{\partial dist}{\partial y_B} \Delta y_B \right| + \left| \frac{\partial dist}{\partial y_A} \Delta y_A \right|$$

6 Ecological assessments via hotspot criteria

Andre Gergs & Thomas Preuss, Silke Claßen, Tido Strauss, Toni Ratte, Udo Hommen

6.1 Introduction

Workpackages 1 and 2 of the GeoRisk project (chapters 4 and 5) aim at predicting concentrations of active substances in edge of field water bodies as a result of drift entries from pesticide applications in permanent crops (orchards, vineyards, hops culture). These PECs (Predicted Environmental Concentrations) might be initial (maximum) PECs or PECs averaged over specific time intervals (PEC_{TWA}). Regardless of the calculation method used, the result is a map of PECs for each of the relevant water body segments (10 – 20 m) for a given plant protection product and a specific use. For the risk assessment and - in consequence – the risk management, these PECs have to be compared with Regulatory Acceptable Concentrations (RAC) derived from standard or higher tier ecotoxicological tests[43].

If the PEC is higher than the RAC, the segment is a potential Risk Management Segment (RMS). However, not every exceedance of a RAC in a segment must indicate an adverse ecological effect:

- The RAC includes a safety factor and thus a PEC above the RAC must not have an effect at all (e.g. the RAC might be the LC_{50} of rainbow trout divided by the standard trigger of 100).
- An effect of an individual organism might not necessarily result in an adverse effect on the population level.
- A spatially restricted effect in a few water body segments might not necessarily correspond to an adverse effect on the population in the field.

Therefore, the hotspot concept was suggested in a former project (UBA F&E project 206 63 402, Schulz et al. 2007) which provides criteria to identify critical aggregations of RMS within a landscape. According to this, a hotspot is a section of a water body with aggregated RMS for which adverse effects on populations cannot be excluded with sufficient confidence.

Two types of hotspot analyses are used in the GeoRisk approach:

1. Within the GeoRisk project, a generic analysis using a virtual pesticide applied with 1000g a.i./ha and a RAC equal to the maximum acceptable concentration according to the current regulation was conducted to identify hotspots in the landscape. For the hotspots it has to be checked at the site, whether the geodata used for the PEC calculations are correct (e.g. distance to the crop, presence of drift reducing vegetation). If in a

[43] The RAC is derived by dividing an ecotoxicological endpoint by a safety factor (corresponding to the triggers used in the TER approach). For example, the RAC can be the acute LC_{50} for fish divided by 100 or an EC_{50} of Algae or *Lemna* divided by 10 or the NOEC or NOEAEC of a mesocosm study divided by a factor < 10 on a case by case basis.

PEC calculation using potentially refined geodata the hotspot criteria are still fulfilled, local risk management options (see chapter 9) along the relevant RMS have to be selected and implemented to reduce the drift entries to a level that unacceptable effects are unlikely (in other words: subsequently no hotspots in the landscape anymore).

2. If these risk management measures have been implemented in the landscape, the final criterion for registration of a specific plant protection product is suggested to the absence of any new hotspot caused by the use of this plant protection product according to Good Agricultural practice. This could be achieved by product specific mitigation measures (e.g. distance measures up to 10 m to surface waters and/or use of drift reducing techniques).

Thus, the definition of the hotspot criteria is a central part of the whole approach in order to ensure the protection of the populations in the edge of field water bodies and by to avoide unnecessary risk mitigation measures.

6.1.1 Generic hotspot criteria (UBA proposal 2007)

Before the start of this project there were a lack on the scientific background on spatial distribution and effects on populations. Therefore a first approach was suggested by UBA (2007) giving the following preliminary hotspot definition which was assumed to be conservative:

A hotspot is given if for 10 or more % of the segments in a stream or ditch section of 1000 m the PEC is higher than the RAC or if at least in one segment a PEC higher than ten times the RAC.

Only the risk segments (PEC > RAC) in such a hotspot are management segments, risk segments outside a hotspot are considered to be tolerable. Therefore, only the entries into the management segments should be managed, not the whole hotspot.

Additionally to the hotspot criteria a method was proposed to define hotspots in a spatial explicit way by using a moving window approach (UBA, 2007). The length of the window has to be set to the 1000 m relevant for a hotspot. Thereby it was explicitly stated that the connectivity of the water body has to be assured, because otherwise no connected populations can be expected. This window should then be moved from the origin of a water body downwards, so that potential risk management segments are located always downstream. This is important since the upstream effects are of major importance within a stream.

This approach includes several assumptions:

- Consideration **of lethal effects** (reduction of abundance) is protective against sublethal effects (e.g. reduction of reproduction by the same amount).
- If PEC > RAC then the whole population in the segment is killed (yes or no lethal response), so the **effect per segment** is 100 %
- 1000 m is a relevant **spatial scale** for a population.
- 10 % reduction of a population is a **tolerable effect**.

The general objective of WP 3 was the development and evaluation of scientifically based criteria for the identification of hotspots. Therefore it was analysed whether the preliminary criteria (namely the effect per segment, the spatial scale and the tolerable effect for populations) of the UBA (2007) can be refined by criteria based on biological traits.

6.1.2 Outline of the steps to derive hotspot criteria

The main questions assigned to WP3 can be summarized as follows:

- How effective are hotspot criteria for the proposed approach?
- Which groups of species or representative species are of interest?
- What is the relevant spatial scale?
- What is the **tolerable effect** for populations undergoing recovery and maybe recolonisation?
- How are **effects per segment** calculated if PEC>RAC?
- How to take short term exposure and chronic effects into account?
- What is a realistic worst-case slope for a concentration-response relationship?
- How will the hotspot criteria be implemented in future risk assessment of plant protection products?

The procedure for the derivation of trait based hotspot criteria is given in Figure 6-1.

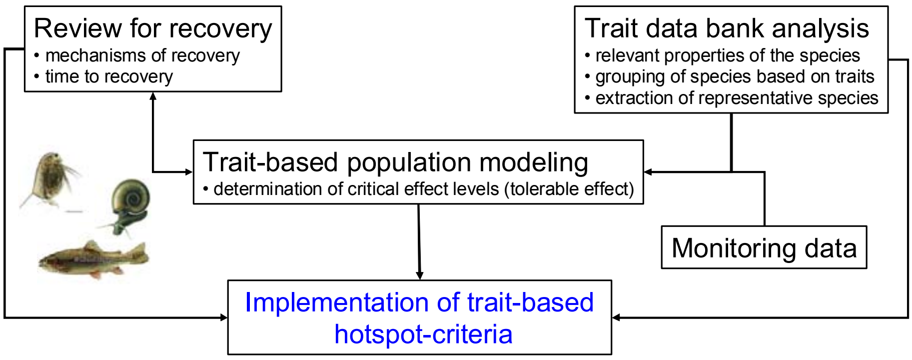

Figure 6-1: Approach to define trait based hotspot criteria

Available data on species composition of edge of field water bodies (small streams, ditches and ponds close to agricultural areas) were used to identify typical community structures. Data bases on biological traits of freshwater organisms provided the data to translate species composition into trait combinations. From this analysis, a set of realistic worst-case species was extracted.

Analysis of case studies of recovery supported the derivation of hotspot criteria by providing information of expected recovery times in the field for different taxa. From this review it was also concluded that recolonisation as additional recovery process can **not** be included in the generic or substance-specific hotspot criteria.

Trait-based population modelling supported the estimation of tolerable effects levels for the different trait combinations.

Concentration-response and time-to-effect relationships from literature analysis were used to define a worst case slope and to give an approximate of how to deal with short term exposure in a conservative, but not unrealistic way.

6.1.3 The trait concept

A trait in this context is a specific property of a species, i.e. a life cycle characteristic, feeding type, mobility type, etc. The basic idea behind the trait concept is that different ecosystems (e.g. small streams in different landscapes, i.e. North and South Europe) might have different species but that these species usually represent typical combinations of traits. Thus, based on traits instead of species comparison and maybe even extrapolation from one (model) ecosystem to another might be possible. Predictable changes in assemblage-wide trait representation have been observed for lotic invertebrate communities along gradients of hydrologic disturbance (Richards and others 1997; Townsend and others 1997; Vieira and others, 2004) and anthropogenic pollution (Charvet et al. 1998, Vierira et al. 2006).

One example for the application of the trait concept is the SPEAR approach (SPEcies At Risk) to evaluate macroinvertebrate monitoring data with respect to the potential effects of pesticides and other organic pollutants (http://systemecology.eu/SPEAR/). For SPEAR, vulnerable species are identified based on four traits: sensitivity to organic chemicals (including pesticides), generation time, time of emergence and dispersal ability (here: ability of up-stream movement within a stream). The proportion of SPEAR individuals or taxa within a community was shown to correlate with the exposure to pesticides in small streams in different regions across Europe (e.g. Liess & von der Ohe 2005, Schäfer et al. 2007).

For more general information on the use of the trait concept in ecotoxicology and environmental risk assessment see also (Usseglio-Polatera et al. 2000, Poff et al. 2006, Baird & van den Brink 2007, Baird et al. 2008, van den Brink et al. 2010). On a recent workshop on trait-based ecological risk assessment (TERA) in 2009 a framework for trait-based assessment in ecotoxicology was established (Rubach et al. 2010). Thereby traits important for external exposure, intrinsic sensitivity and population sustainability were identified and weighted by their importance and data availability. It was concluded that despite the challenges that remain, trait-based approaches have the potential to enable ecotoxicologists to develop a more mechanistic approach to understand the reasons for differences in the vulnerability of populations to toxicants. Ultimately this enhancement of understanding should allow an improvement to extrapolate between species, thereby providing more effect risk assessment methodologies.

Within GeoRisk, the trait concept was used to differentiate the above mentioned generic hotspot criteria to allow the consideration of different species. One important aspect to be considered with the hotspot criteria is the population sustainability, the potential of populations to recover from short term disturbance. In principle, recovery can occur from inside the disturbed system by reproduction of survivors or from insensitive resting stages (autogenic recovery), or via recolonisation from outside the system (allogenic recovery). Therefore, traits related to demography (e.g. voltinism, life span, clutch size) and recolonisation (e.g.mobility or dispersal potential) appeared to be the appropriate traits for the derivation of hotspot criteria.

6.2 Refined hotspot criteria

6.2.1 Sensitivity analysis for the generic UBA criteria

The preliminary hotspot criteria (UBA 2007) are based on three quantitative assumptions to be refined in the trait based criteria:

1. the relevant spatial scale: 1000 m in the generic criteria
2. the acceptable proportion of segments with PEC exceeding the RAC (generically 10 %) and
3. the effect size assumed if the RAC is exceeded (100 % in the generic criterion)

The product of 2 and 3 relates to the tolerable effect on the population (here a reduction of abundance of 10 % is suggested to be acceptable).

For the determination of the acceptable effect magnitude both the intrinsic recovery and the recolonisation have to be considered. However, if the RAC is based on a NOEAEC from a mesocosm study including recovery, it has to be made sure in the final approach, that recovery is not considered twice (in the RAC and in the hotspot criteria).

Using the dataset "Bodensee" of the JKI a first analysis on the **sensitivity** of the number of risk management segments (RMS) against changes of these three parameters was conducted.

The analysis of the dataset revealed that a consistent network of water bodies is lacking. Only 89 % of the water bodies and 36 % of the segments were associated with water bodies of a maximal length of 40 segments which equals 1000 m, the length of the moving window as proposed by the UBA (Figure 6-2).

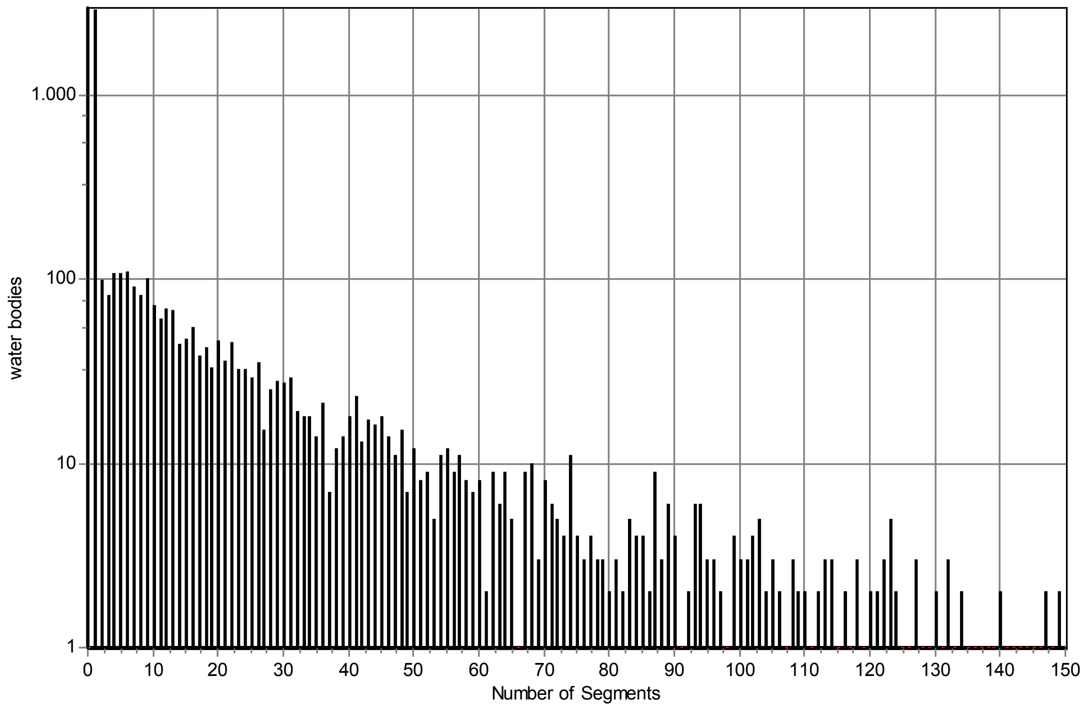

Figure 6-2: Frequency of streams and ditches of different length (number of segments) for a JKI dataset of the Lake Constance region

The results of this sensitivity analysis show that the length of the considered stream (or ditch) section (length of the moving window) has only low effect (above a minimum length) on the number of hotspots (Figure 6-3). Thus, it is probably sufficient to use the same window length for all taxa respectively trait groups. However, the effect of the window length on the number of RMS depends also on the landscape structure and the connectivity of the water bodies in the dataset.

Most important for the resulting number of RMS were the assumptions on the tolerable total effect and the dose-response relationship (which effect is assumed if the PEC is higher than the RAC).

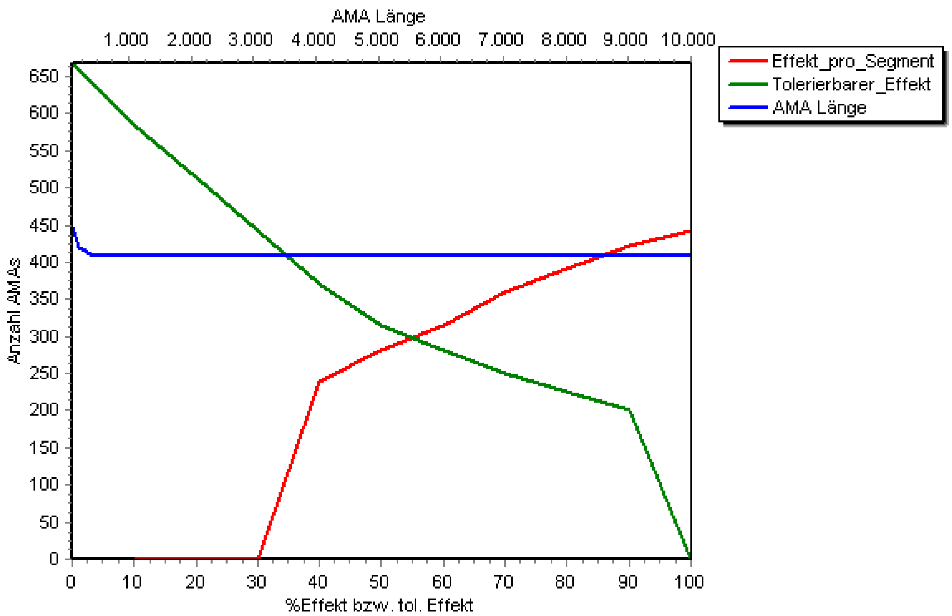

Figure 6-3: Sensitivity analysis for a JKI dataset of the Lake Constance region: number or RMS (Anzahl AMAs in the figure) depending on the assumed effect per segment if the RAC is exceeded, (‘ % Effekt bzw. tol. Effekt’) the assumed tolerable effect per segment and the length section analysed (moving window, ‘AMA Länge’)

A second sensitivity analysis conducted by JKI, figured out that the window length might have an influence at higher tolerable effects (Figure 6-4). Since it was figured out that only 10 % effect can be tolerated for populations under generic realistic worst-case assumptions (chapter 6.2.7.10), the window length remain an insensitive parameter for hot-spot identification.

Conclusion:

Sensitivity analysis indicated that the total length of hotspots was not very sensitive for the spatial scale considered to be relevant for the population. Additionally it was found in the literature that even populations with low dispersal abilities are able to manage distances of 1 km. On the other side, a larger window size for the hotspot criteria will always reduce the number of management segments.

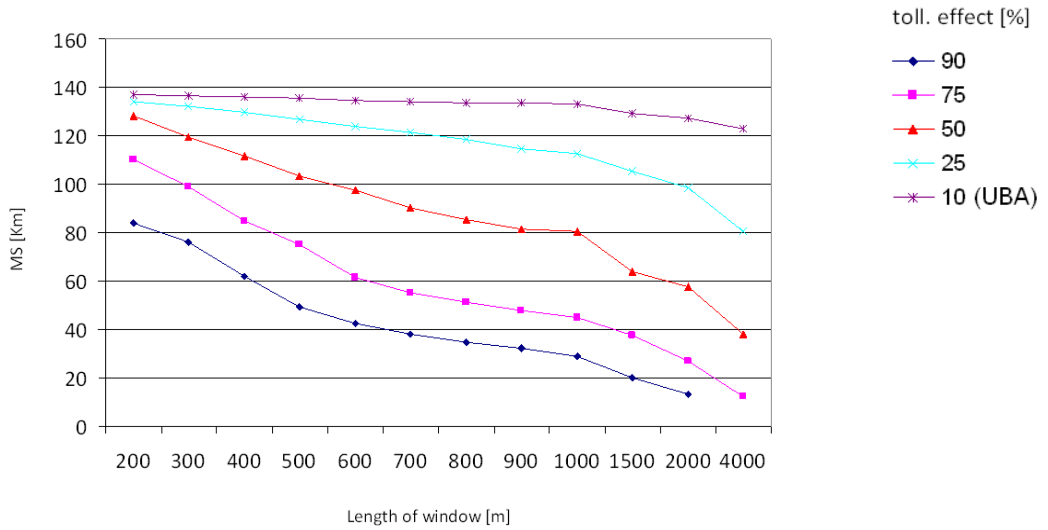

Figure 6-4: Influence of tolerable effect and length of sliding window on MS length: Example of the fruit growing region Lake Constance using the static exposure model (provided by B. Golla)

6.2.2 Recovery and recolonisation processes – a review

In order to provide background information on recovery and recolonisation processes observed after disturbance of freshwater communities, a literature review was conducted. This work has been summarized in a manuscript included in Appendix D. Thus, only a summary is given here.

The ISI web of science and related article functions of further online tools were scanned for papers related to recovery or recolonisation and to stream, ditches, ponds, micro- and mesocosms, or enclosures as well as different types of stressors (Figure 6-5) in the aquatic environment. On the base of title and abstract, a total of 471 and 152 articles were collected for lotic and lentic systems, respectively. Case studies including at least one endpoint of recovery were selected based on the availability of data for system characteristics, stressor, effects and recovery processes. By applying these criteria, the selection included 150 publications in lotic systems and 76 articles in lentic systems, resulting in a total number of 148 cases studies and 908 endpoints of recovery.

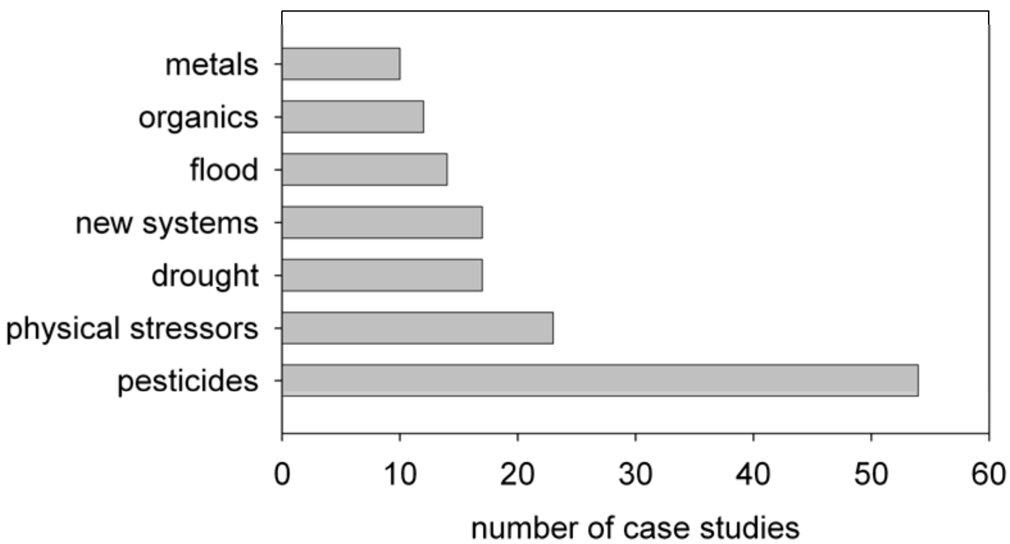

Figure 6-5: Number of cases in five categories of stressors included in the study

Recovery processes were analysed for different taxonomic groups. The majority of endpoints were identified for macroinvertebrates. In comparison to previous reviews (e.g. Niemi et al 1990) data for zooplankton, algae, and aquatic macrophytes increased in the recent years. Within the group of macroinvertebrates most of the data was available for Diptera, Ephemeroptera, Coleoptera, Trichoptera, and Heteroptera. General patterns of recovery were extracted from the literature for different taxonomic groups and examples of case studies are given. Times to recovery differed within taxonomic groups and ecosystems (Figure 6-6, Figure 6-7), resulting in characteristic patterns of post disturbance succession.

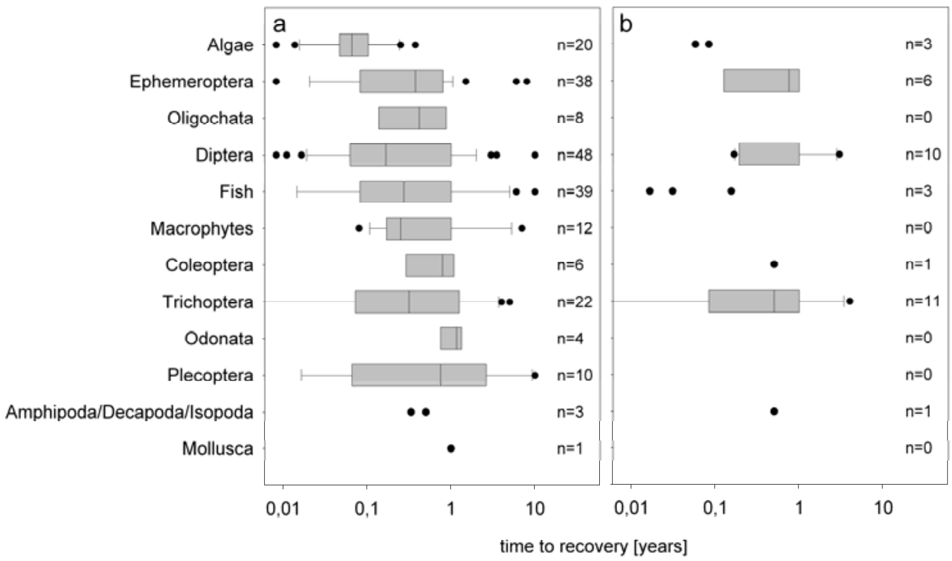

Figure 6-6: Time to recovery of selected taxonomic groups in lotic systems after stress: (a) all stressors included in the study, (b) pesticide. Boxes represent median, inner and outer quartile. Taxonomic groups sorted in descending order by their outer quartile.

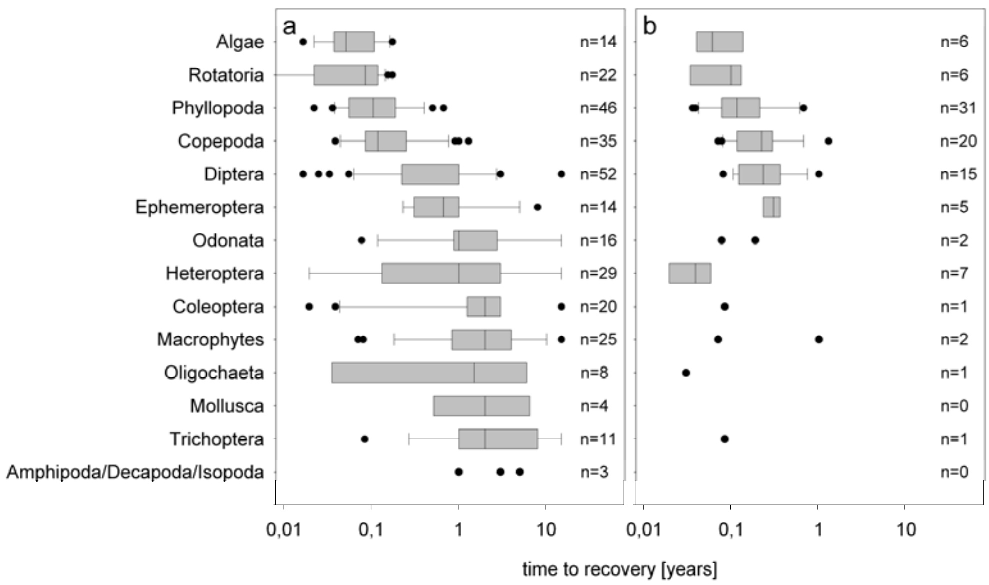

Figure 6-7: Time to recovery of selected taxonomic groups in systems after stress: (a) all stressors included in the study, (b) pesticide. Boxes represent median, inner and outer quartile. Taxonomic groups sorted in descending order by their outer quartile.

Zooplankton and phytoplankton recovered in nearly all cases within one year even after pronounced effects. Fast recovery was observed in algae followed by Ephemeroptera and Diptera populations independently of ecosystem type. Lowest recovery potentials were found for populations of Trichoptera, Odonata, Plecoptera and Crustacea in lotic systems. In systems benthic crustaceans, populations of Trichoptera, Mollusca and Coleoptera did mostly not recover within one year. In general recovery of lotic macroinvertebrates appeared to proceed faster than recovery in systems likely due to drift of organisms from undisturbed upstream reaches.

If the analysis is restricted to pesticides only, the low amount of available data does not allow any interpretation of recovery for specific taxonomic groups, but the overall pattern seen in the whole dataset is mostly reflected.

Interestingly, recovery of endpoints related to biodiversity like taxa richness, community composition and diversity indices lasted longer than endpoints for single species (Figure 6-8). This astonishing fact is obviously due to methodological reasons, since recovery is mostly measured for high abundant or selected species (e.g. Baetidae or Ephemeroptera), whereas in the analysis of taxa richness, community composition or diversity indices also the low abundant species are taken into account. The latter also considers the equal distribution of species, and therefore the recovery of the low abundant groups. This example shows that overall recovery of ecosystems might be longer lasting than can be expected from literature studies observing single species abundances.

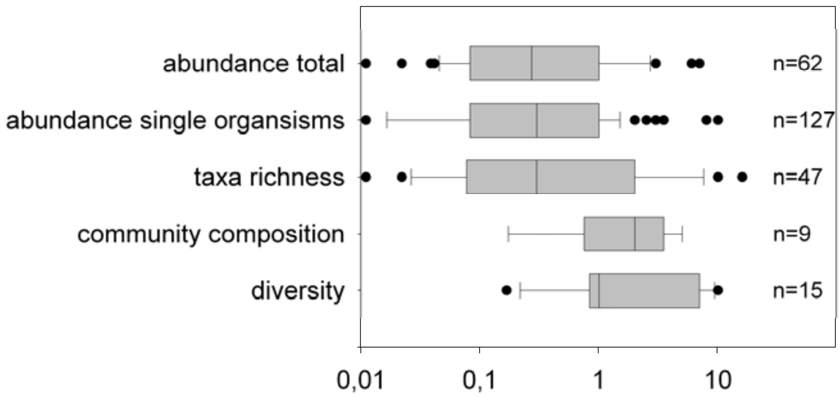

Figure 6-8: Time to recovery of community measures in lotic macroinvertebrates. Taxa richness includes recovery in overall macroinvertebrates or selected taxonomic subgroups, community composition includes principal response curves and indices of similarity; diversity integrates different diversity indices

Furthermore factors driving recovery processes independently of taxa are discussed in our review. According to Wallace (1990) recovery times of macroinvertebrates after pesticide stress are depending on the following factors:

- Magnitude of the original introduction, toxicity and extend of continued use of the pesticide
- Spatial scale of disturbance
- Persistence of the pesticide
- Timing of contamination in relation to the life history stage

- Position within the drainage network

One important fact concluded in previous reviews (Niemi et al. 1990, Yount & Niemi 1990, Wallace 1990, Mackay 1992) is the dependence of recovery from certain physical characteristics of the ecosystem. In general, recovery in lotic systems was faster if undisturbed sites were present upstream or refugial areas present in the affected reaches (Cuffney et al. 1984). Thus, drift is seen as one of the dominating recolonisation and recovery pathways within lotic systems. In most cases, it could not be figured out whether recovery was autogenic or allogenic. However, for several taxa with only one reproduction period per year (e.g. fish) the cases of fast recovery indicate that active movement was the main pathway of recovery while for taxa with high population growth rates (algae, zooplankton, some multivoltine macroinvertebrates) recovery can likely be explained by population growth alone.

In the literature, stressors for aquatic ecosystems are grouped as pulse or press disturbance. Whereas recovery after pulse exposure might be fast, recovery after press disturbance is longer lasting. As defined by Yount et al. (1990) spills of non-persistent chemicals (e.g. pesticides currently in use) typify a pulse disturbance, whereas long-term pollution or clear-cutting of a forested watershed typifies press disturbances. This pattern was also found in the current study by plotting affected macroinvertebrate endpoints as a function of time to recovery (Figure 6-9). Two distinct groups can be extracted from time of recovery data. Within the first group represented by four types of stressors - physical disturbance, flood, drought and pesticides -, recovery within one year was observed in around 80 % of the macroinvertebrate endpoints. In a second group, organics, metals and constructed wetlands are clustered. For this group more than 50 % of the macroinvertebrate endpoints take more than one year to recover.

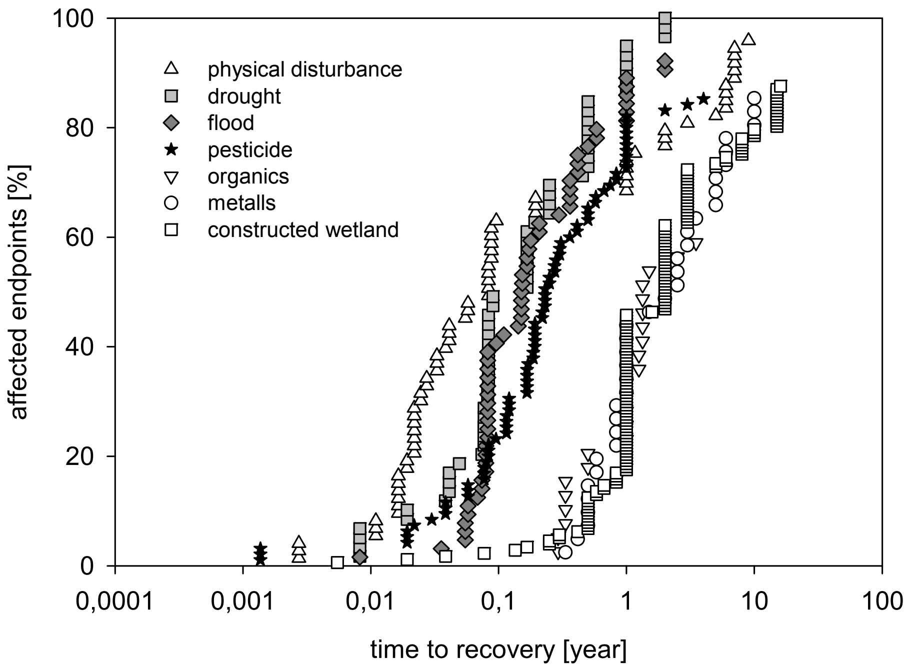

Figure 6-9: Cumulative frequency of observed recovery times for all macroinvertebrates and endpoints after stress in lotic and lentic systems. Dots represent data derived from the literature, grouped by type of stressor.

In contrast, case studies investigating multiple pulse exposure or long-term effects of the chronic use implied that pesticide application can become a press disturbance in aquatic ecosystems.

Because additional stressors multiple applications and mixture toxicity are not considered in the case studies focussing on single stressors, recovery predictions based on these case studies might be too optimistic.

For prediction of recovery in risk assessment of pesticides three main factor groups have to be taken into account: (1) stressor related factors, (2) species related factors, and (3) factors related to habitat and landscape structure. Whereas stressor related factors are the basis of current risk assessment, species related factors are only marginally involved, but trait based approaches might help in future to consider this. In system related factors, the spatial heterogeneity of landscapes (e.g. presence, position and quality of refugial areas) have to be taken into account for an adequate prediction of recovery. The data included in this review clearly indicates that lotic and lentic systems differ in their potential for recovery and thus have to be separated within risk assessment. Whereas approaches to model recovery in dependence of factors related to stressor and species already exist, no approach is available to deal with habitat diversity (refuges) and landscape structure (presence of sources for recolonisation). Whenever recolonisation is considered in pesticide risk assessment, these obstacles have to be removed.

6.2.3 Identification of realistic worst case species using the trait concept

6.2.3.1 Aim

To conduct a hotspot analysis external exposure, intrinsic sensitivity as well as population sustainability (van Straalen 1993) has to be taken into account to evaluate whether a risk from a given exposure can be identified. Since all three properties are species dependent identification of realistic worst case species has to be conducted to come up with a conservative assessment of risk from spatial exposure patterns. For simplification and to be conservative, traits of species which might reduce the external exposure were ignored. In the following chapter realistic worst case species were identified from monitoring data of edge of field water bodies using a trait-based approach.

6.2.3.2 Definition of relevant ecological traits

For the purpose here, identification of ecologically critical aggregations of water body segments with PECs above the RAC, the following traits were considered as relevant, concordant to a recently published framework for trait-based assessment in ecotoxicology (Rubach et al. 2010) and with respect to population sustainability including demography and recolonisation potential:

- Life cycle traits determining the population growth rate (e.g. survival rates, development times, voltinism, clutch size)
- Presence of insensitive stages (e.g. resting stages)
- Mobility und dispersal distances

Other traits might also be important for possible indirect effects, but were not included yet: oxygen deficit tolerance, macrophytes as substrate, habitat or food. Habitat preference and distribution and frequency in Germany were not considered because the trait based approach is per se not geo-referenced, all groups representing trait combinations are assumed to be present in each water body and thus, potentially exposed. Definition of traits in general and specific traits used in this approach can be found in the framework for trait-based assessment in ecotoxicology (Rubach et al. 2010).

Any individual organism itself possesses multiple traits, and traits are not necessarily concordant with phylogeny (Poff et al. 2006), e.g. not all members of the Plecoptera are predators. A set of traits can be identified for a species (or higher taxon), this specific trait set is defined as a taxon's function trait niche (FTN), which is analogous to the ecological niche of a species, i.e. the total of attributes defining its unique relation to its environment and other species. (Poff et al. 2006). FTNs are non-random because they have been structured by evolution and, accordingly, closely related taxa are more likely to have similar FTNs than distantly related taxa (Poff et al. 2006). Therefore it cannot be the aim of this WP to build up one artificial worst case trait group, by combining worst case traits. In contrast, a group of species showing critical traits will be selected due to their relevance for edge of field water bodies and low recovery potential.

A first rough grouping of aquatic species was conducted within a WP3 workshop based on the following set of criteria:

- The groups cover the main ecological functions (primary producers, primary and secondary consumers, decomposers)
- The groups cover taxa found to be sensitive in the former registration of plant protection products: plants, arthropods, fish as usually tested but also non-arthropod invertebrates as for example molluscs.
- The groups respectively the representing species is relevant for edge of field water bodies.
- If possible, basic autecological data are available to allow modelling of population dynamics for estimation of recovery times.
- Different types of life cycles and dispersal mechanisms are covered.
- Finally, a practicable number of groups respectively reference species can be selected for derivation specific hotspot criteria.

From these criteria a draft set of trait groups (equal to realistic worst case species) could be constructed satisfying the needs of ecological risk assessment. These theoretical trait groups are given in Table 6-1 and were used as a starting point prior to further refinement within this WP. Two options were discussed within the work group: Selection of representative trait combinations should be based (1) on available trait data bases or (2) alternatively on data available for community structure in typical edge of field water bodies. It was decided to follow the second option and subsequently to use trait data bases to gather information on representative species.

Table 6-1: Preliminary trait based grouping of fresh water organisms to demonstrate the principal approach

Trait group	Trophic level / feeding type	Sens.	Reproduction	Generat. / a	Mobility	Dispersal	Example
Algae	primary producers	H	+++	n	low	passive	*Scenedesmus*
Makrophytes	primary producers	H	+ / ++ (vegetativ)	1 (irrelevant)	no	passive	*Myriophyllum*
Zooplankton	filterfeeder	I, F	++	2 - n	low	passive	*Daphnia, Cyclops*
Crustacean	detritus feeders	I, F	++	1-2	medium	drift, movement	*Asellus, Gammarus*
Insects 1	primary consumers	I, F	++	1-2	high	drift, flight, movement	*Baetis, Ephemera*
Insects 2	predators	F, M	+		high	drift, flight, movement	Notonectidae, Odonata,
Molluscs / non arthrop.	grazers	I, F	+	1- n	low	passive	*Lymnaea, Radix*
Small fish	secundary consumers	I, F	++	1	high	active / passive	Stickleback, minnow
Large fish	tertiary consumsers	I, F	+	1	high	active	Trout, pearch

6.2.3.3 Available trait data bases / trait concepts

Several data bases providing species or taxon related trait information were identified and used within the WP:

- “Bayernliste“ (Schmedje & Colling 1996, MS-EXCEL file of the BLfW on freshwater macrofauna)
- Fauna Aquatica Austriaca (Moog 1995, 2002)
- ASTERICS: macroinvertebrates for use under water framework directive, 13000 taxa, http://www.fliessgewaesserbewertung.de/
- POND FX: http://ipmnet.org/PondFX/pondlife_main.htm
- Tachet – French book on invertebrates
- Lotic Invertebrate Traits (North America) – digital data base
- SPEAR: http://systemecology.eu/SPEAR/
- Storefish – digital database on 80 European fish species covering > 60 traits

Further useful links include

- http://www.ephemeroptera.de/
- http://www.trichoptera-rp.de/
- http://www.mollusken-nrw.de/
- http://www.plecoptera.de/
- http://www.benthos.org/index.cfm
- http://www.fba.org.uk/
- http://www.freshwaterlife.org/eco_db_home.jsp

These databases covered all possible traits, but are not consistent regarding the species, traits nor incorporated data, data handling and data coding. Most traits within the databases did not regard the given questions and were therefore ignored, e.g. saprobic index, feeding type or habitat preferences.

The „Bayernliste“and the Fauna Aquatica Austriaca (Moog 1995, 2002) include ecological traits, not relevant for the given question, for nearly 2000 macrozoobenthos taxa or higher entities (genera, family, etc.). The same ecological traits are covered by the ASTERICS database, but for nearly 13 000 macrozoobenthos taxa.

The SPEAR-database was developed to identify vulnerable species based on four traits: sensitivity to organics, generation time, time of emergence and dispersal ability (here: ability of upstream movement within a stream), it covers nearly 2000 macrozoobenthos taxa.

Only three databases contain a high number of various traits, including ecological, morphological and life cycle traits, the POND-FX, TACHET and the Lotic Invertebrate Traits (North America) – digital database. The last three databases were used for the above mentioned further analysis.

6.2.3.4 Analysis of monitoring datasets to identify representative species

In order to identify representative species relevant for edge of field water bodies monitoring datasets were screened. The following macroinvertebrate datasets were identified to provide data on community structure:

- streams around Braunschweig: Pantel (2002), Wogram (1996, 2010)
- ditches in the Altes Land near Hamburg: Schäfers et al (2006)
- streams in the hops region Hallertau: Classen et al. (unbublished)

The outcome of this task was a list of species available at edge of field water bodies in Germany. The list of representative species was linked to the trait data bases which resulted in trait groups representative for German edge of field water bodies. Based on these traits a choice of representative species (realistic worst case species) was selected to be used in the derivation of hotspot criteria.

Among the factors governing the recovery of freshwater invertebrate species after toxic perturbation one important factor is its life history (chapter 6.2.5). Species have different life-history characteristics, such as lifespan, time to first reproduction, and number of offspring produced over a lifetime (Stark et al 2004) which can be summarised as generation time or voltinism. Another mechanism that facilitates recovery or recolonisation is the ability of immigration (Sherrat et al 1999). The dispersal ability of a species can be attributed as a trait related to immigration.

Factors that influence the resistance of freshwater invertebrate species to toxic stress are their sensitivity to the toxicant and the presence of aquatic stages during pesticide application.

The observation that species with different life-history strategies may react differently to stressors resulted in advanced concepts of pesticide risk assessment. For example Lies and von der Ohe (2005) employed the four traits generation time, dispersal ability, sensitivity (relative to *Daphnia magna*) and presence of aquatic stages to identify species at risk (SPEAR). In general, vulnerable species are assumed to exhibit low dispersal ability and long generation time. Species with a full aquatic life cycle and species with emerging adults exhibiting a larval development of > 1 year can per se be assumed to be present during PSM application periods. Consequently, species showing slow dispersal and a time to reproduction of two years (semivoltine organisms) can be referred to as potential realistic worst case species. In order to identify realistic worst case species, in a first step the trait composition related to dispersal and voltinism of macroinvertebrate communities were analysed and afterwards the population sustainability of species was identified.

For a general overview on trait distributions species data available for Braunschweig, Altes Land and Hallertau were related to the Tachet and POND FX trait data bases. Preferably information on genus or species level (Tachet) was used. The lack of data, due to missing species in the Tachet data base, was filled by using information on family level (PondFx).

Although different species[44] occurred in the three agriculture landscapes, the analysis demonstrated that the dispersal of traits related to dispersal ability or voltinism are nearly equally distributed across the sampling regions (Figure 6-10). Species reproducing once a year (univoltine) were found most frequently (54-67 %) whereas semivoltine species (6-12 %) were collected less frequently. This finding is in range of the relative distribution of different voltinism categories as found in macroinvertebrate communities sampled in other German regions and in Dutch clay ditches (Wogram 2010, Brock et al. 2010). In these evaluated studies univoltine and semivoltine species comprised 52-74 % and 3-16 % of all macroinvertebrates with known voltinism. A large proportion of the species sampled in Braunschweig, Altes Land and Hallertau showed a high ability of dispersal (52-64 %) whereas 20-24 % of the species had a low ability of dispersal.

[44] With respect to the monitoring data sets, the term ‘species’ is used pragmatically for the lowest taxanomic level a specific organism could be identified to. Thus, this ‘pseudospecies’ can be a real species, or e.g. a genus, family etc. if further differentiation was not done in the study.

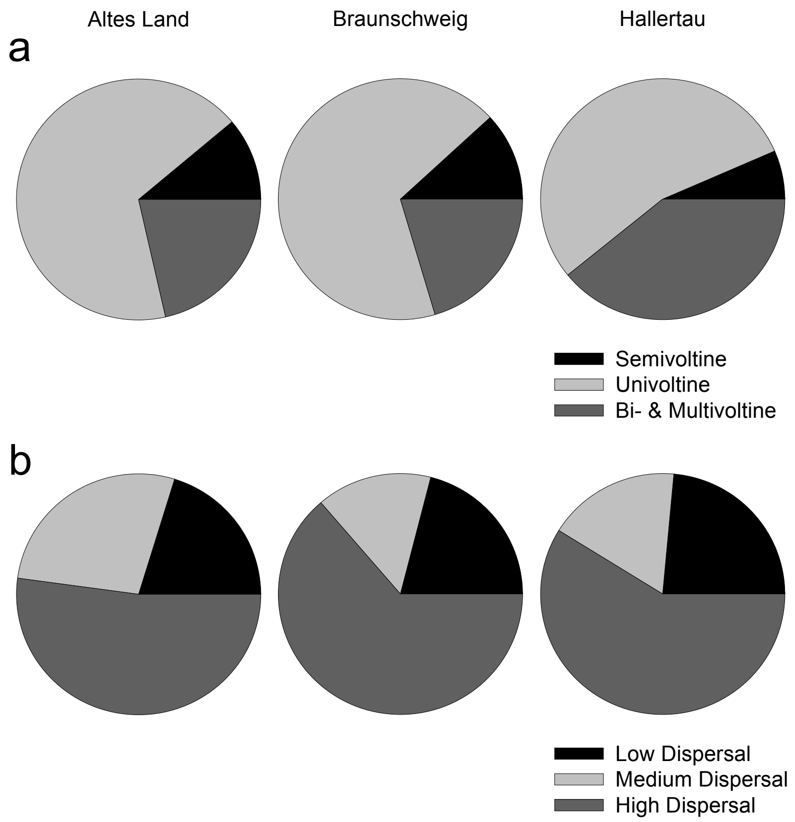

Figure 6-10: Relative distribution of traits, related to voltinism (a) dispersal ability (b), within aquatic macro-invertebrate species of steams and ditches in three German agricultural areas

Furthermore the analysis revealed that the two traits, dispersal ability and voltinism, are not randomly combined. Only multivoltine species show high dispersal whereas low dispersal ability was found to be exclusive to semivoltine taxa (Figure 6-11).

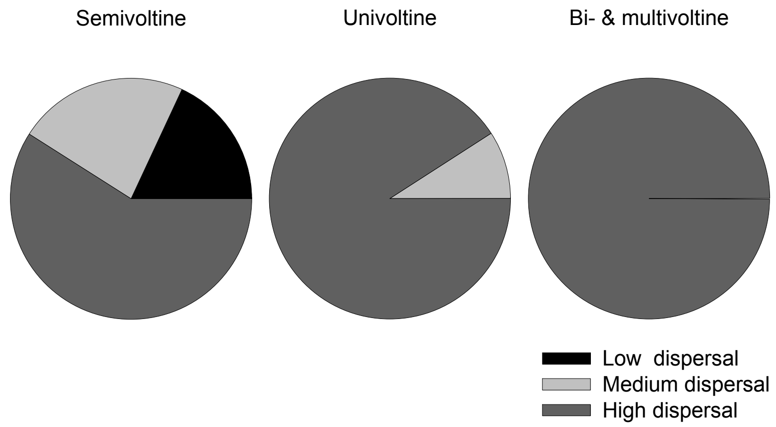

Figure 6-11: Relative distribution of dispersal ability within three groups of species traits related to voltinism of the three German agricultural areas

In order to identify species with low population sustainability (those sensitive against stressors due to their life strategies), datasets were divided into those from potential reference sites with low or medium exposure and those from potentially affected sites with high exposure, similar to the classification suggested for the Altes Land by Schäfers et al. (2006). Since only reference data was available for the Braunschweig region, data from Altes Land and Hallertau was involved in this second analysis. Species were classified and grouped according to their occurrence in reference sites. Species exclusively appearing in sites with low or medium exposure or being found in low numbers within affected sites compared to reference, were classified as potentially sensitive species (Group a). Within the second group, those species were clustered which were found at all sites or mainly at sites with high potential for exposure (Group b). A list of classified species is given in (Table 6-2). The subdivision of species as done for the two groups is supported when comparing the number of taxa declared as SPEAR (Figure 6-12). According to the SPEAR data base 30 % of the species in group b can be classified as species at risk, whereas 5 % of species of group b are at risk.

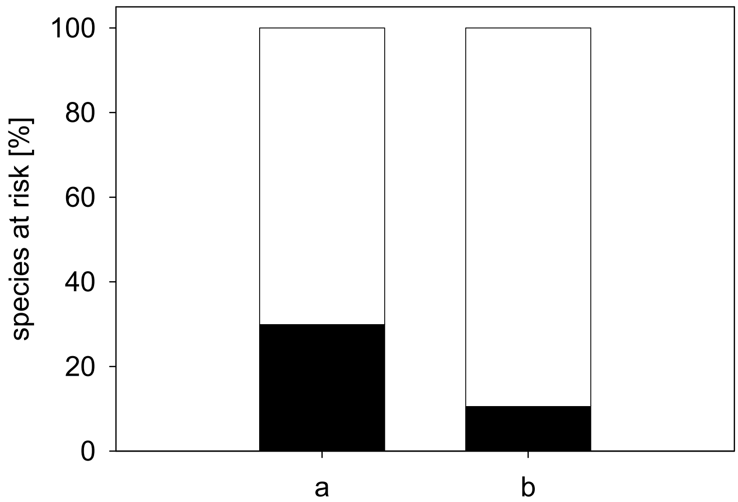

Figure 6-12: Percentage of aquatic species at risk according to the SPEAR database in the two German agricultural areas Altes Land and Hallertau. Macro-invertebrates were grouped by their occurrence within potential reference sites (a) and their additional occurrence in potentially polluted sites (b).

Table 6-2: List of potentially sensitive species (a) and species mainly found at sites with high potential for exposure (b)

Group a	**Group b**
Potential sensitive taxa	Potential in-sensitive taxa
Altes Land	
Acilius sulcatus	*Anisus vortex*
Acroloxus lacustris	*Coenagrion puella/pulchellum*
Aeshna grandis	*Dugesia lugubris et polychroa*
Agabus undulatus	*Erpobdella nigricollis*
Anax imperator	*Erpobdella octoculata*
Anisus vorticulus	*Glossiphonia complanata*
Anodonta cygnea	*Glossiphonia heteroclita*
Asellus aquaticus	*Haliplus ssp*
Bithynia leachii	*Helobdella stagnalis*
Caenis robusta	*Lumbriculus variegatus*
Cloeon dipterum	*Notonecta glauca*
Colymbetes fuscus	*Physa fontinalis*
Dero digitata	*Planorbarius corneus*
Hydrophilus aterrimus	*Planorbis carinatus*
Laccobius ssp	*Planorbis planorbis*
Limnephilus stigma	*Radix ovata*
Nais pseudoobtusa	*Rhynchelmis limosella*
Pisidium casertanum	*Sigara ssp*
Ranatra linearis	*Stylaria lacustris*
Triaenodes bicolor	
Hallertau	
Agabus didymus	*Tanytarsini ssp*
Macroplea ssp	*Eiseniella tetraeda*
Hydrophilidae ssp	*Tanypodinae ssp*
Calopteryx virgo	*Radix balthica*
Tipula platytipula	*Hydropsyche angustipennis*
Simulium ssp	*Gammarus roeseli*
Dixa ssp	*Erpobdella octoculata*
Tabanidae ssp	*Tipula maxima*
Pseudolimnophila ssp	*Galba truncatula*
Halesus tesselatus	*Prodiamesa olivacea*
Psychodinae ssp	Oligochaeta ssp
Eloeophila ssp	Chironomini ssp
	Orthocladinae ssp

The comparison of the relative distribution of traits related to voltinism within aquatic macroinvertebrate taxa revealed no differences between the two agricultural areas (Figure 6-13). Furthermore, the number of bi- and multivoltine species, i.e. species with a time to reproduction ≤

0.5 years, is only slightly increased in sites with high potential for exposure compared to references. In contrast, the distribution of dispersal ability was found to be different between reference and affected sites (Figure 6-13). The number of species with low dispersal ability is notably increased in affected compared to reference sites.

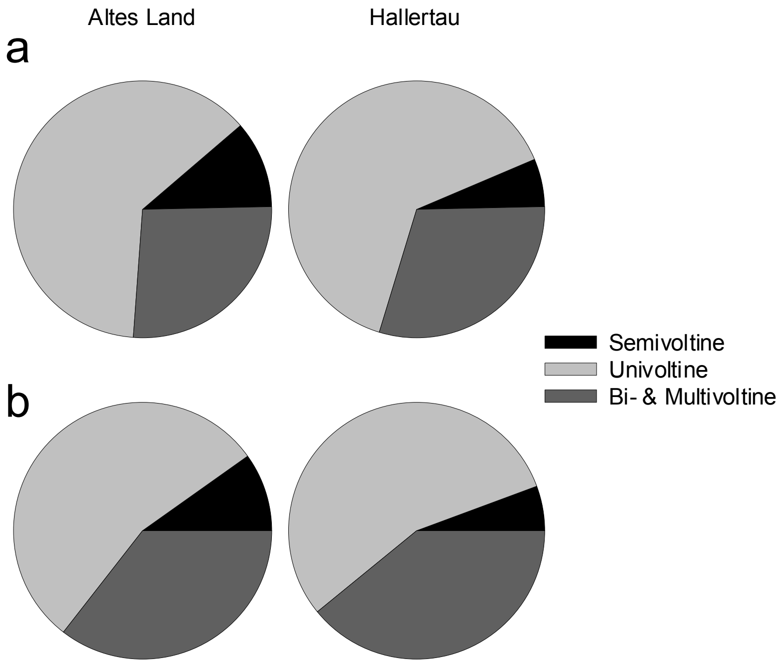

Figure 6-13: Relative distribution of traits related to voltinism within aquatic macro-invertebrate taxa of steams and ditches in two German agricultural areas. Macro-invertebrates were grouped by their occurrence within potential reference sites (a) and their additional occurrence in potentially polluted sites (b).

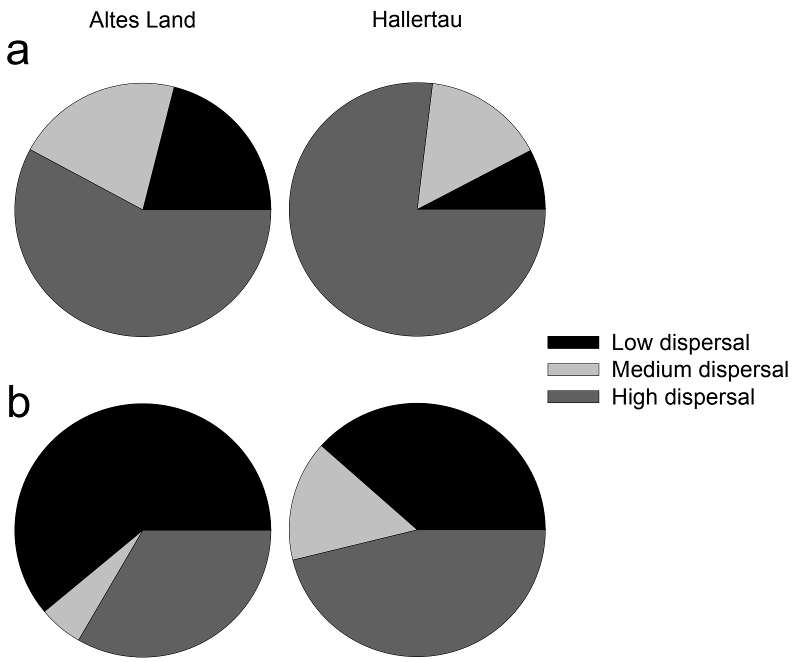

Figure 6-14: Relative distribution of traits related to dispersal ability within aquatic macroinvertebrate taxa of steams and ditches in two German agricultural areas. Macroinvertebrates were grouped by their occurrence within potential reference sites (a) and their additional occurrence in potentially polluted sites (a).

In a next step, life cycle strategies as suggested by Verberk et al. (2008) of two groups of species were compared. A list of species classified into 13 trade based life-history strategies is available from Verberk (2008). Life-history strategies were outlined based on species traits, their interrelations and functional implications. Since "…different strategies representing different solutions to particular ecological problems…" (Verberk et al. 2008), different life cycle strategies should be present in sites with different potential for exposure.

Less than 10 % of species sampled at the agricultural areas in Germany are listed in the compilation of species by Verberk (2008). This might be due to different geographical regions and different habitats being included in the studies. Nevertheless, the analysis demonstrates that there is no overlap in life strategies between the two groups of species, except for life strategy D1, representing species with strong dispersal ability (Table 6-3). This might again reveal the accuracy of grouping species in reference and potentially polluted sites.

Table 6-3: Life strategies (LF) of species found at sites with low (group a) and high potential for exposure (group b)

Taxon	T. Group	LF	Description
Group a			
Acilius canaliculatus	Coleoptera	D1	Short development time, strong dispersal
Agabus undulatus	Coleoptera	S3	Short juvenile development time
Dero digitata	Oligochata	R4	Low age at 1st reproduction, no active flight
Limnephilus stigma	Trichoptera	S2	Short growth period and resistant stages
Nais ssp.	Oligochata	R4	Low age at 1st reproduction, no active flight
Ranatra linearis	Heteroptera	R1	Sequential reproduction, active dispersal
Triaenodes bicolor	Trichoptera	D1	Short development time, strong dispersal
Group b			
Asellus aquaticus	Isopoda	R3	Sequential reproduction, brood care, no active flight
Cloeon dipterum	Diptera	D2	Large clutch size, strong dispersal
Dugesia sp	Seriata	S4	High per capita investment, no active flight
Eiseniella tetraedra	Oligochaeta	T2	High tolerance, no active flight
Erbobdella ocultulata	Hirudinea	R2	Seq. reproduction, many small eggs, no active flight
Erbobdella testacea	Hirudinea	R2	Seq. reproduction, many small eggs, no active flight
Glossiphonia complanata	Hirudinea	R2	Seq. reproduction, many small eggs, no active flight
Glossiphonia heteroclita	Hirudinea	S4	High per capita investment, no active flight
Notonecta glauca	Hemiptera	D1	Short development time, strong dispersal
Rhynchelmis limosella	Lumbriculida	S4	High per capita investment, no active flight
Sigara ssp	Hemiptera	D3	Low age at 1st reproduction, strong dispersal

Since the concept of life strategies did not allow identifying species integrating worst case trade combinations (low dispersal and long life cycle) from the current monitoring data, a literature search on life-cycle traits was conducted to achieve more detailed ecological information on species showing critical trait combinations.

From the group of species mainly occurring at reference sites, trait combinations of univoltine or semivoltine as well as low or high dispersal ability were selected. This approach led to the identification of realistic worst case species from different interesting taxonomic groups. The literature search clarified that the generalisation as done on family level for life cycle strategy and dispersal ability within the PondFX database is not always transferable to single species since the specification of a trait might differ between species of a family (e.g. the dispersal ability) or even within a species (e.g. voltinism depends on latitude as shown for Odonata by Corbet et al (2006)). Furthermore, the literature search on characteristics of particular species led to a refinement of trait classifications, shown in Table 6-4.

Table 6-4: Selected potential realistic worst case species from monitoring datasets.

Taxon	T. group	Voltinism	Dispersal ability
Anodonta cygnea	Unionidae	> semivoltine	low
Dero digitata	Oligochata	uni- to semivoltine (+ asexuell)	low
Nais pseudoobtusa	Oligochata	uni- to semivoltine (+ asexuell)	low
Aeshna grandis	Odonata	semivoltine	high
Anax imperator	Odonata	uni- to semivoltine	high
Calopteryx virgo	Odonata	semi- to univoltine	low
Acroloxus lacustris	Acroloxidae	uni- to multivoltine	low
Anisus vorticulus	Planorbidae	uni- to multivoltine	low
Bithynia leachii	Bithyniidae	(semi-), uni- or multivoltin	low

In a next step species classified as low dispersal taxa were sorted by their reproductive strategy (Figure 6-15). It is expected that species reproducing intermittently throughout their lives (polycyclic) generally show a fast recovery after perturbation, whereas species that have long time to reproduction and reproduce only once in their lifetime (semelparous) or species that produce offspring in successive, annual cycles and survive over multiple seasons (iteroparous) generally show lower population sustainability.

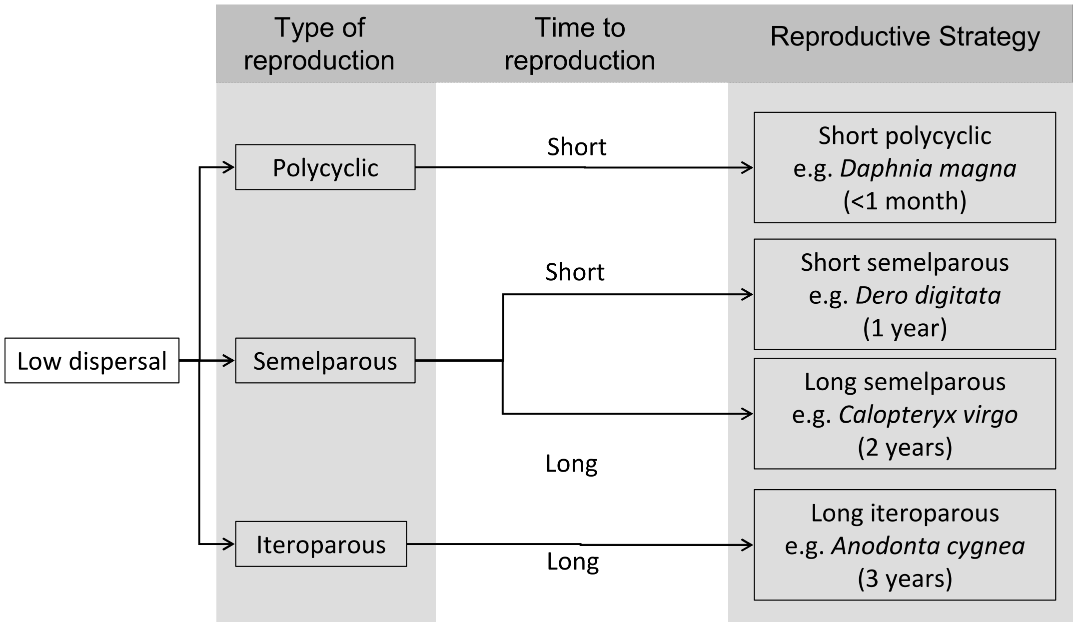

Figure 6-15: Definition of reproductive strategies based on the type of reproduction and time to reproduction, for each group a species and its development time is given as an example.

In summary, representative species were selected from a reference list based on combinations of traits related to voltinism and dispersal ability. Species with aquatic larval development time ≥ 1 year were considered present during application of plant protection products. Iteroparous and semelparous species with low dispersal ability are suggested to show low population sustainability. For the derivation of hotspot criteria, selecting species with a low ability of dispersal appears to be a pragmatic but conservative solution.

Applying these considerations, it turned out that *Anodonta cygnea*, *Calopteryx virgo* and *Dero digitata* are promising candidates to reflect realistic worst case species which can be used in a modelling approach for the identification of hotspot criteria. Life cycle characteristics of the three species selected are briefly outlined below.

The freshwater clam *Anodonta cygnea* prefers lentic habitats, however it also can be found in streams in lower number compared to other genera of large clams, e.g. *Unio*, (Ravera and Sprocati 1997). Mollusca of the genera *Anodonta* generally reach maximum shell length of 15 cm within ~10 years (Bauer 1994), and start to reproduce at a shell length of ~3 cm (Bauer 1994), equals to 3-5 years in age (Mueller 1996). In *A. cygnea* glochidial (larval) developments starts in late summer to autumn, glochidia can be found in the gills (for release) in late autumn and winter (Aldridge, 1999). The glochidia of Unionidae generally attach to a host fish to pass through their parasitic stage. In particular, glochidia of *A. cygnea* require only a short period, compared to other Unionidae species, to complete development into the young mussel (Bauer 1994). Life cycle strategy of *Anodonta cygnea* is not listed in the compilation as done by Verberk (2008). Applying his criteria the species could best be classified as R2 (sequential reproduction, many small eggs, no active flight), a life strategy found in potential affected sites within the monitoring data. Due to its low sensitivity to insecticides *A. cygnea* is listed as a species not at risk (Liess and von der Ohe 2005).

Larvae of the damselfly *Calopteryx virgo* were found to prefer small sized running waters. In intermediate latitudes *C. virgo* generally shows a semivoltine life cycle (Corbet 2005). In Odonata species, the synchronization of emergence within a certain period and its seasonal placement tend to be consistent for a species in a given climatic situation (Corbet 1980). *Calopteryx* adults usually emerge in July and depose eggs in the water column attached to plants (Gibbons and Pain 1992). Oviposition events can be observed during a period of several days to few weeks. Although it is believed that Odonata species are generally showing high dispersal ability, mark-and-recapture experiments showed that less than 5 % of adult *Calopteryx* migrate beyond a distance of 500 m from origin (Stettmer 1996). *Calopteryx* species are not listed in the lifecycle classification of Verberk. Life cycles of other Zygoptera were mostly described by T1 (high tolerance, active dispersal) or D2 (large clutch size and strong dispersal); however this classification might not hold true for *C. virgo.* Like many other damselflies, *C. virgo* is considered to be a species at risk within the SPEAR classification.

Naidida worms are primarily linked to the epilithon and periphyton (Cellot 1998). However, *Dero digitata* is also considered to be a swimming organism (Drewes and Fourtner 1993, Cellot 1998). The worm undergoes both, sexual and asexual reproduction. In field studies, asexual periodic reproduction is evident in almost all naidid species. Sexual reproduction co-occurred at the time of greatest abundance (Parish 1981). Within trait data bases *Dero* is listed as uni- to semivoltine species as a result of irregularly observed sexual reproduction. In contrast, seasonal peaks due to asexual reproduction can be found reproducibly (Smith 1986). Within a species the seasonal placement of peaks might depend on climate situation. Spring peaks as well as autumn peaks were reported in *Dero digitata* (Parish 1981, Smith 1986). Within the classification of Verberk, *Dero digitata* is considerd to exhibit a R4-type of life cycle strategy (low age at first reproduction, no active flight). *D. digitata* is not listed in the SPEAR data base since only few toxicity data is available (e.g. Mischke et al 2001).

Mollusca, Odonata and Oligichaeta species were among those taxa which were found in the literature review (chapter 6.2.2) to recover only slowly after disturbance which was mainly attributed to their low potential for recolonisation. Other taxa that also turned out to recover slowly like Plecoptera, Heteroptera, Coleoptera and Trichoptera taxa were either not resident in the monitored agricultural areas due to their habitat requirements or were possibly not affected.

6.2.3.5 Conclusion

Although different species occurred in the three agriculture landscapes, the analysis demonstrated that the dispersal of traits related to dispersal ability or voltinism is equally distributed across the sampling regions. This finding is in range of the relative distribution of different voltinism categories as found within macroinvertebrate communities sampled other German regions and in Dutch clay ditches. The trait dispersal indicates that the trait based approach can be used for the given question and results in comparable outcome for different regions. Furthermore the analysis revealed that the two traits, dispersal ability and voltinism, are not randomly combined. Multivoltine species only show high dispersal whereas low dispersal ability was found to be exclusive for semivoltine taxa. From the group of species mainly occurring at reference sites trait combinations of univoltine or semivoltine and low or high dispersal ability were selected. This approach allowed the identification of realistic worst case species from different interesting taxonomic groups, whereby iteroparous and semelparous species with low dispersal ability and long life span or voltinism are suggested to show low population sustainability. Applying these considerations, it revealed that *Anodonta cygnea*, *Calopteryx virgo* and *Dero digitata* are promising candidates to reflect realistic worst case species.

6.2.4 Defining the spatial scale

The sensitivity analysis of the generic UBA criteria (chapter 6.2.1) indicated that the reduction of hotspots was not sensitive for the spatial scale. Additionally it was demonstrated that even populations with low dispersal abilities manage to travel distances of 1 km (chapter 6.2.2 and 6.2.3).

A larger window size for the hotspot criteria will always reduce the number of management segments.

Therefore, 1 km is used as the relevant spatial scale in the hotspot criteria.

Nevertheless the sensitivity analysis was conducted on the ATKIS dataset and thus, it might be possible that the low sensitivity against this parameter is only attributed to the high amount of disconnected water bodies. Therefore the importance of the length of the moving window should be re-evaluated if a more convenience network of water bodies will be used later on.

6.2.5 Defining tolerable effects

6.2.5.1 Aim

For a generic hotspot analysis aimed at identifying water body segments where risk mitigation would have the highest benefit for the protection of the populations, realistic worst case assumptions on dose-response relationships must be used. The ecological vulnerability and thus, the tolerable effect of populations depend on the following properties

1. Tolerance of the population against effects on the individual (elasticity, see Forbes et al. 2009): The same effect on one life-cycle trait can have different effect on the population growth rates at different species;
2. Intrinsic population growth rate determining the time to reach the pre-disturbance abundance again;
3. Presence of insensitive resting stages to allow recovery even after (preliminary) extinction and without recolonisation;
4. Recolonisation potential depending on the stages, mobility and dispersal distances, but also on landscape properties (presence of sources for recolonisation)

The objective of the following section is to define the magnitude of effect which can be tolerated by a population considered these properties. For the hotspot criteria, an effect will be considered to be tolerable if annual application over a period of 10 years would not result in a significant reduction of the population abundance. For species with life expectations of several years also the stability of the age structure should be considered.

6.2.5.2 State of the art modelling approaches

The effects of chemicals and pesticides on populations in the field depend not only on the exposure and the toxicity, but additionally on other factors such as life history characteristics (e.g. generation time, fecundity), population structure, density dependence, the timing of exposure, community structure and occurrence of other stressors (Barnthouse et al. 2007, Seitz & Ratte 1991, Solomon et al. 2008). Ecological modelling represents an excellent tool to explore the importance and interaction of such factors and it is assumed that the effects on populations can be predicted (Forbes et al. 2009, Thorbek et al. 2009). Since in environmental risk assessment of chemicals and pesticides the protection goal is usually related to the population or the community level (see e.g. Hommen et al. 2010) , ecological modelling has the potential to provide more ecologically relevant endpoints than a risk assessment relying on measured endpoints derived from single species tests. Such endpoints might be the time to recovery as well as extinction probability calculated on population level. Therefore efforts for establishing mechanistic modelling in environmental risk assessment for chemicals and plant protection products are currently being evolved (Grimm et al. 2009, Preuss et al. 2009). Ecological models of all types are not only used to study fundamental ecological processes, but also have practical applications. Examples include models supporting forest management (Huth & Tietjen 2007, Porte & Bartelink 2002) , fishery (Pauly et al. 2000), and biological conservation (Lindenmayer et al. 1995). Simulation models are useful for extrapolating from laboratory, or semi-field studies to field situations (Lopes et al. 2005, Naito et al. 2002, Van den Brink et al. 2006) or to predict recovery time (Barnthouse 2004, Van den Brink et al. 2007).

Different modelling approaches can be applied to estimate effects and recovery time of population, from very simple models using logistic growth curves over compartment and matrix models to individual-based models (Smolke 2010), whereby it is not clarified yet which model approach is suitable for the given question. Due to the time limitations within the project advanced model approaches like individual-based models were disregarded, regardless of their high potential to answer the given questions (Preuss et al. 2009, 2010, van den Brink 2007) and it was agreed within the working group that a simple population model based on the identified traits should be used instead.

A first simple approach focussing mainly on autogenic recovery via population growth of survivors is described in Barnthouse (2004) using the Verhulst equation for logistic population growth.

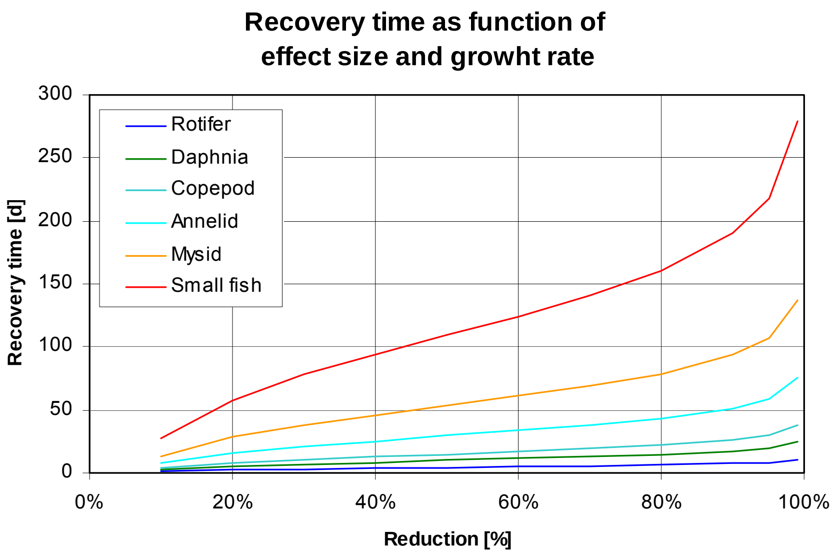

Figure 6-16: Recovery time for different aquatic taxa depending on the size of effect (here: reduction of abundance); modified from Barnthouse (2004).

This models allows to estimate recovery times for species with continuous reproduction and thus, it is probably suitable to determine tolerable effects for plankton organisms, *Lemna* and maybe some other fast growing populations.

Calow et al (1997) used a simple two stage model with five parameters for elasticity analysis which might be used as a simple trait-based population model for the extrapolation to the population level. However, the model is based on the assumption of unlimited (intrinsic) population growth and therefore might overestimate recovery times.

For the use in GeoRisk, the following spatial explicit modelling approach was suggested:

The model includes the main mechanisms of recolonisation: the organismic drift (Figure 6-17):

- Passive downstream transport depends on flow velocity and biological trait.
- Mobility: short-distance movements, depends on flow velocity and biological trait.
- Upstream flight/walk (drift compensation), depends on biological trait and environment.
- General dispersal (long-distance movements, passive transport), depends on biological trait and environment.

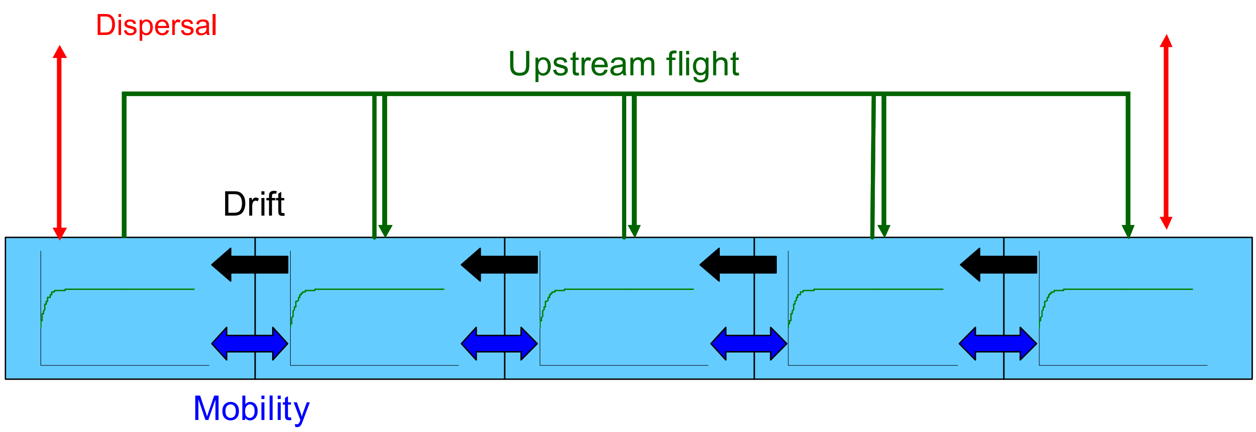

Figure 6-17: Scheme of the different mechanisms driving recolonisation in stream sections

Due to technical restrictions in the GIS-tool, recolonisation could not be implemented in a geo-referenced way within the project. Since recolonisation depends - beside on the mobility of the species - strongly on the landscape, the connectivity of the water bodies as well as on the availability of refuges (Chapter 6.2.2) it can also not be included in the generic tool.

Since all three identified realistic worst-case species show a complex life-cycle with reproduction only once a year and several years to reach maturity model approaches, as the two approaches described above, based on the assumption of continuous reproduction are not suitable.

6.2.5.3 Modelling approach

A stage-based model with discrete reproduction was developed. The stage-based model is given by Equation 6-1. This model describes the changes in abundance over time as a function of mortality and development rate. Thereby a density dependence of the mortality rate was assumed, at higher abundance mortality increases and at lower abundances mortality decreases. The abundance of a specific stage is increased by the number of animals from the previous stage and reduced by the animals which finished their development. The model used in this approach was set to three stages. Therefore no development from a lower stage to stage 1 and no development in further stages than stage 3 was simulated, so the development rates dev0 and dev3 were set to 0. Stage 3 is the reproductive state and reproduces at a specific time of reproduction (TOR) into stage 1, given by Equation 6-3. If semelparous reproduction was found and the adults died at reproduction, the abundance of stage 3 was set at TOR to the abundance of stage 2 and the abundance of stage 2 were set to 0 (Equation 6-2).

Equation 6-1:

$$\frac{\Delta S_{n,i}}{\Delta t} = \left(S_{n,i-1} \times dev_{i-1}\right) - \left(S_{n,i} \times dev_i\right) - S_{n,i} \times m_i \times \frac{S_{n,i}}{K}$$

$S_{n,i}$: abundance of stage i and segment (n)

m_i: mortality rate [d-1] of stage i

K: capacity

dev_i: development rate [d-1] of stage i

Equation 6-2:

If semelparous reproduction and t=TOR $\begin{matrix} S_{n,3} = S_{n,2} \\ S_{n,2} = 0 \end{matrix}$

Equation 6-3:

If t=TOR $S_{n,1} = S_{n,1} + 0.5 \times S_{n,2} \times Offspring$

Effects were implemented to occur only at the time of application (TOA) and the abundance of segment n is reduced by the effect in this segment (Equation 6-4). It was assumed that the plant protection products act equally on all stages.

Equation 6-4:

IF t = TOA $S_{n,i} = S_{n,i} \times (1 - effect_n)$

Dispersal ability of the species was only included as drift in the current modelling approach. It was concluded from the literature review that the scientific base for including recolonisation, i.e. immigration from other water bodies, and upstream movement in the current approach is not given, additionally all selected realistic worst-case species show low dispersal potential. Simulation of drift was included, because it was one of the main factors triggering recovery in the literature review. Drift between the segments was implemented by using Equation 6-5. A drift rate (d) is allocated to each stage and the abundance of segment n is reduced by the drift rate and at the same time the abundance is increased by drift from the upstream segment.

Equation 6-5:

$$\frac{\Delta S_{n,i}}{\Delta t} = S_{n-1,i} \times d_i - S_{n,i} \times d_i$$

d_i: drift rate [d^{-1}] of stage i

In this model approach acute toxicity can be implemented by reducing the number of individuals and chronic effects by reducing the growth rate. The time of exposure is essential. If only drift is taken into account, a single or multiple short peak is expected. Therefore it seems to be over-protective to use chronic toxicity datasets. If chronic toxicity has to be taken into account the model needs exposure patterns over time, which will not be generated by the tool developed within this project. Additionally complex exposure patterns might result in higher or lower effects than standard toxicity test conducted under constant exposure (Preuss et al. 2009, Reinert et al. 2002). This topic has to be postponed to future activities.

In the project, it was not possible to implement the model in GIS but it was applied to a virtual stream with variable number of segments. The outcome of the model represents the tolerable effect for populations of the representative species identified in 0 which allows stable population development over 10 years of yearly application. Uncertainty of the model results regarding tolerable effect sizes was considered qualitatively by the use of worst-case assumptions if possible (e.g. use of the lowest number of offspring reported, ignoring recolinization, effects on all stages, and timing of application). However, on the other side competition with other species was not considered, which might affect recovery, and only the effects of one application of one pesticide were considered.

6.2.5.4 Model calibration

The stage-based model with discrete reproduction was calibrated for the identified realistic worst-case species. The resulting parameters are given in Table 6-5. The model was initialised by 100 individuals of stage 2 and 3 and 0 individuals of stage 1; then the model was allowed running 3 years under control conditions to establish the population. Afterwards the control and treatment scenario was calculated. Capacities of 100 individuals in each stage were assumed. The development rate, offspring per female, time of reproduction and mortality rate of stage 3 were calibrated to literature data. In contrast the mortality rates for stage 1 and 2 were calibrated to state that maintained a stable population over the years.

The trait group **short semelparous** was calibrated by means of life-history characteristics of the realistic worst-case species *Dero digitata,* an oligochaete which shows a complex life-cycle pattern, in which asexual reproduction is dominating and sexual reproduction occurs seasonally in a distinct short time interval. During the abundance peak in autumn triggered by sexual reproduction field densities of 900 individuals per m^2 can be found (Smith 1986). For simplification the asexual reproduction was ignored and only the population dynamics triggering sexual reproduction was implemented in the model. Therefore the offspring per female were set to 10 000 so that a peak up to 10 000 individuals per segment (approximately 400 individuals per m^2) were reached in the autumn peak. The other parameters were calibrated in a way that the population dynamics in the field (Smith 1986) was reflected by the model.

The trait group **long semelparous** was calibrated by means of life-history characteristics of the realistic worst-case species *Calopteryx virgo*, a damselfly which shows semelparous reproduction strategy. Here, the adults die at reproduction and are only able to reproduce within a short period in their life. Their voltinism can range from univoltin to semivoltin (up to 3 years in Finland) depending on latitude (Corbet 2006), for which reason a two years life-span was implemented in the model. Therefore, stage 1 represents eggs, stage 2 first year larvae and stage 3 second year larvae. In so doing it is assumed that larvae of stage 3 emerge and become adults which deposit their eggs into the water without changes in the abundance. To reproduce the two year semelparous life-cycle dev_1 was set to 1, so that all eggs become immediately first year larvae. Dev_2 was set to 0, so the first year larvae become second year larvae at TOR. Zygopterans to which *Calopteryx virgo* belongs, lay 100 to 400 eggs per episode and some species up to 1800 in 4-14 days (Corbet 1980). Therefore, offspring was set to the minimum of 100 eggs per female. Mortality rates were calibrated in a manner that stable populations were achieved and the first year larvae were more dominant than the second year larvae.

The trait group **long iteroparous** was calibrated by means of life-history characteristics of the realistic worst-case species *Anodonta spec.*, a molluscan clam which shows iteroparous reproduction strategy. Here the adults are able to produce several broods during their lifespan. For the clam stage 1 equals to veliger larvae released once a year by adult clams, stage 2 describes juvenile clams and stage 3 adult clams. A continuous development from stage 1 over stage 2 to stage 3 was implemented, in which the larvae need half a year to become juvenile clams and the juveniles need 3 years to reach the adult stage. The juvenile development was calculated from the length at first reproduction 40 mm (Bauer 1994) and the length over age regression (Mueller 1996); then the larvae development was calibrated. The mortality rate of the adult clams (m_3) was set to 0.0004 d^{-1} which equals a life-span of 6.8 years, the mortality rate of the juvenile clam equals a life-span of 0.5 years and for the veliger larvae a life-span of 0.06 years was calibrated. The reported overall life-span in literature can be up to 10 years (Mueller 1996). The offspring per female were set 3100, representing the lowest value found in literature (Bauer 1994). These parameters result in the population dynamics given in the Figure 6-18.

Table 6-5: Parameters for realistic worst-case species (representative species)

	Description	Unit	*Calopteryx* Long semelparous	*Anodonta* Long iteroparous	*Dero* Short semelparous
Dev_1	Development rate	d^{-1}	1	0.0055	0.4
Dev_2	Development rate	d^{-1}	0	0.0009	0
m_1	Mortality rate	d^{-1}	0.01	0.045	0.001
m_2	Mortality rate	d^{-1}	0.00149	0.006	0.1
m_3	Mortality rate	d^{-1}	0.001	0.0004	1
K	Capacity		100	100	100
Offspring	Offspring per female		100	3100	10000
Semelparous reproduction			Yes	No	Yes
TOR	Time of reproduction	d	70	160	250

The different population dynamics emerging from the model are shown in Fig. 6-18 over 2 years. All three populations show a discrete annual reproduction, which results in a suddenly high increase of stage 1 abundance leading also to a high increase in total abundance. Population dynamics of the long iteroparous and the long semelparous trait group are characterized by a nearly constant level of stage 3 animals and a fluctuating number of stage 2 animals. Population of the short semelparous trait group consist mainly of stage 2 animals, whereas stage 1 and stage 3 animals occure only within a short time interval during the reproduction. For comparison measured population dynamics for the short semelparous trait group (D. dignitata) are shown in Fig. 6-19.

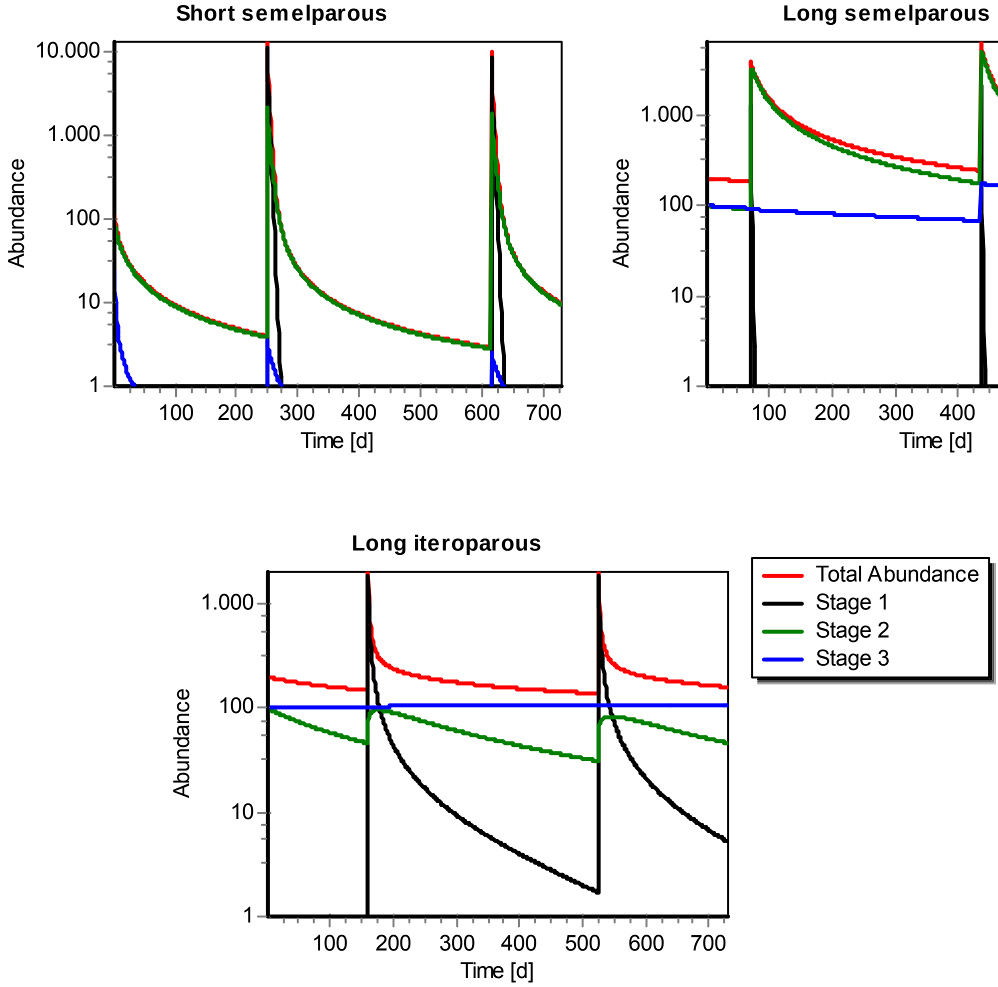

Figure 6-18: Population dynamics for the three trait groups over two years

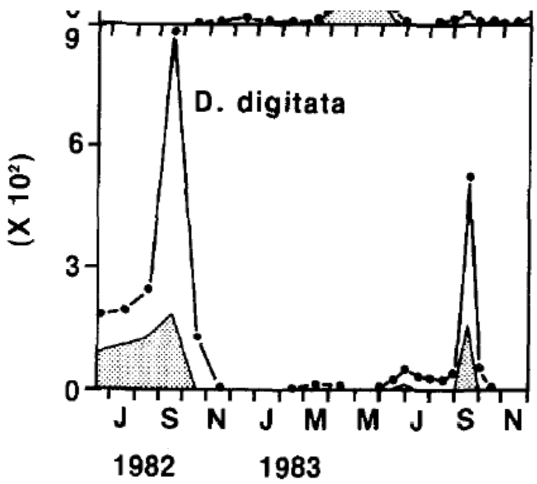

Figure 6-19: Measured population dynamics for the oligochaet *D. digitata* over two years (Smith 1986). The figure was copied from Smith 1986 and demonstrates the seasonal changes of *D. digitata* in a bog stream (worms m^{-2}). The dark area under curve represents number of worms undergoing asexual reproduction.

6.2.5.5 Application of toxicants to a single model population

Effects on population dynamics were tested first of all for a single population. Therefore the model was used only for one segment; dispersal by drift was switched off and a single peak, equal to 50 % effect, at day 100 of a year was applied yearly. The effects on the population dynamics are shown in Figure 6-20, the deviation to control within one year in Figure 6-21 and the effects on the populations over 10 years with yearly application is shown in Figure 6-22. To calculate effects over 10 years three endpoints were evaluated, the abundance at time of reproduction (TOR), time of application (TOA) and the mean abundance within one year, in which the TOR means a timepoint related to the life-cycle of the trait group, the TOA a timepoint related to the toxicant.

The population dynamics show the effects of the toxicant by reducing the abundance at each stage. At the day of application for the long semelparous trait group only stage 2 and stage 3 were present, for the short semelparous trait group only stage 2 and for the long iteroparous trait group all three stages. From the population dynamics it could already be derived that the effects at the second year were more pronounced in the long iteroparous trait group, but not in the long and short semelparous trait group. This pattern is more easily to observe in Figure 6-21 in which the deviation to control for total abundance is plotted. From this figure it becomes obvious that for the short semelparous trait group full recovery can be observed within one year and the effects within the second year are equal to first year. The long semelparous trait group was not able to recover totally within one year, but after the reproductive day (day 70) until day of application (day 100) the population recovered. Afterwards the application of the second year results in the same effect as the application from the first year. In contrast the long iteroparous trait group was not able to recover before the application of the second year and therefore effects on populations increased compared to control.

The above mentioned trend becomes even more obvious if simulations are conducted over 10 years with yearly application. Here even after 10 years no effects could be observed on the short semelparous trait group. For the long iteroparous trait group the effect on yearly mean abundance is stable over the time. The effect based on abundance at time of reproduction is increasing in the second year, because in the first year the application was after the day of reproduction and therefore no effect can be observed with this endpoint. For the long iteroparous trait group the effect was increasing over the time, whereas the increase is more pronounced in the first four years and constant within the last three years. The abundance at TOR was a more sensitive endpoint than the mean abundance or the abundance at TOA.

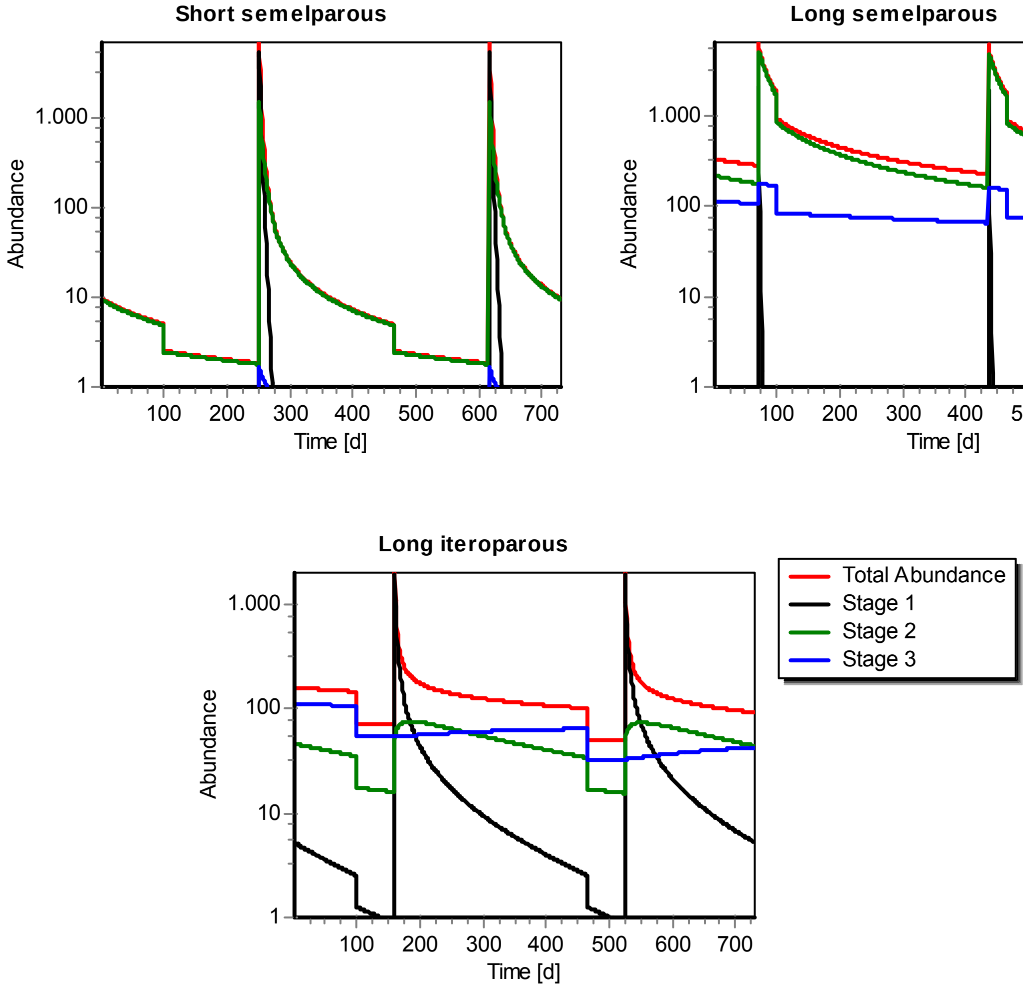

Figure 6-20: Population dynamics at application of a toxicant for the three trait groups over two years. The toxicant was applied at day 100 and 465 at a concentration equal to 50 % effect.

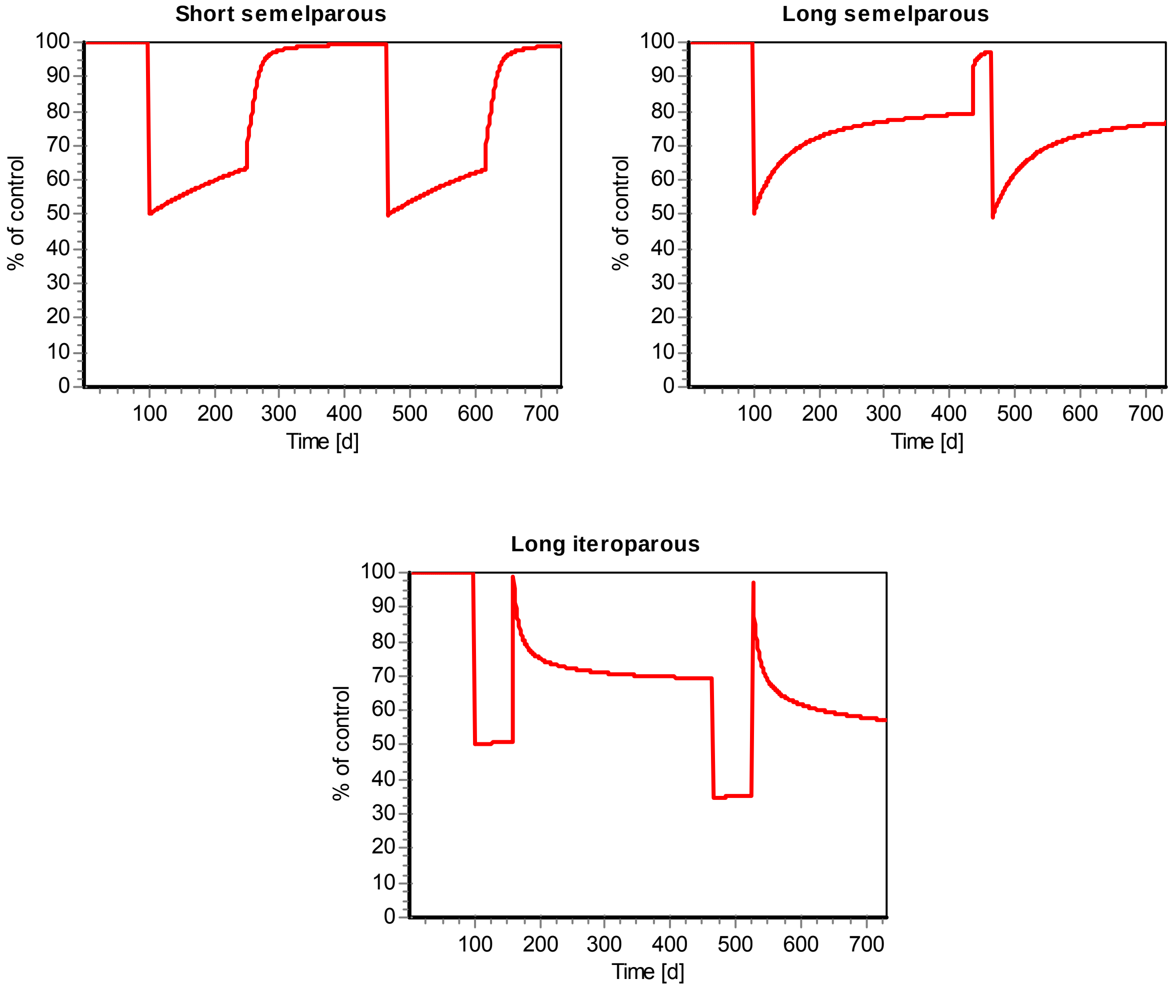

Figure 6-21: Effects resulting from application of a toxicant for three trait groups over two years. The toxicant was applied at day 100 and 465 at a concentration equal to 50 % effect.

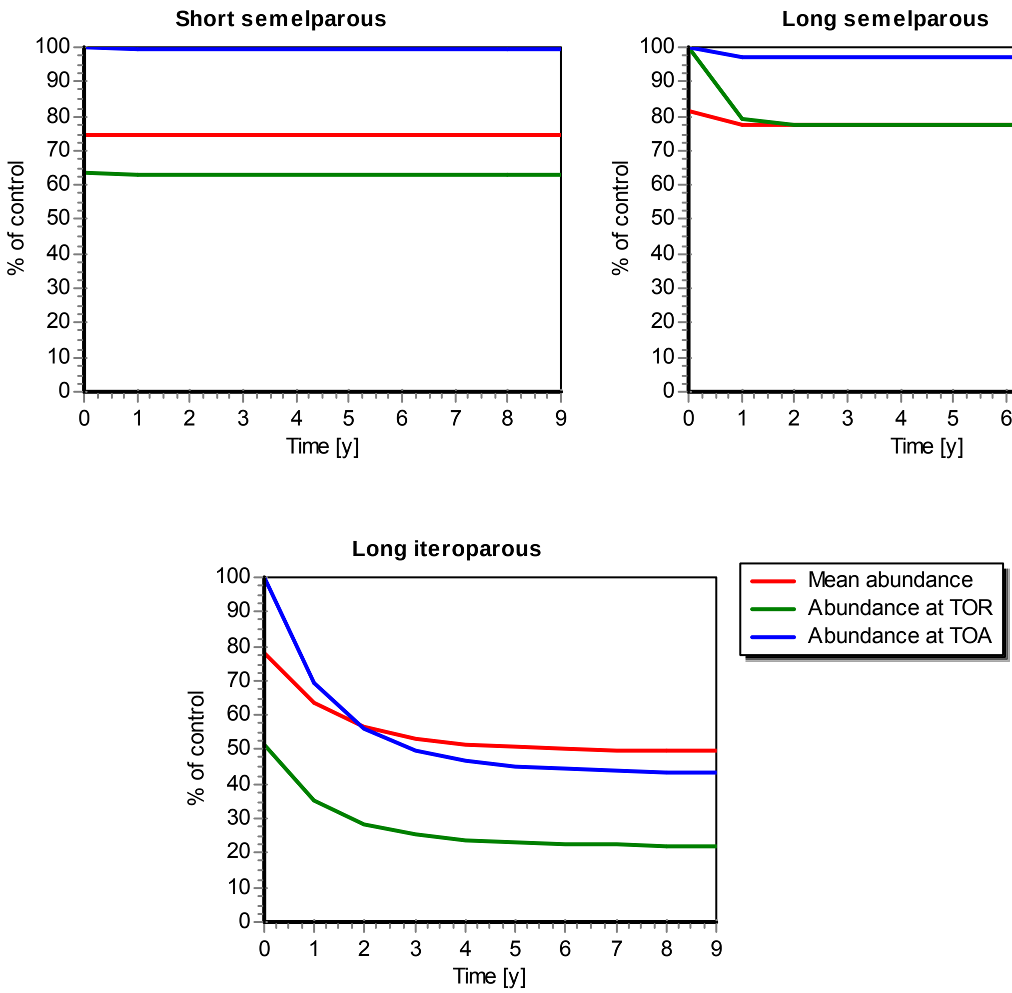

Figure 6-22: Effects resulting from application of a toxicant for the three trait groups over ten years Three endpoints were calculated the abundance at the time of reproduction (TOR), the abundance at time of application (TOA) and the mean abundance over the year. The toxicant was applied yearly at day 100 at a concentration equal to 50 % mortality.

6.2.5.6 Impact of time of application

The influence on the effects of the day of application on the three realistic worst case species was tested using a single peak of a toxicant at a concentration resulting in 50 % mortality over 10 years with yearly application. The results for all three evaluated endpoints are plotted in Figure 6-23. For the long iteroparous and semelparous trait group the abundance at the time of reproduction (TOR) and abundance at time of application (TOA) were more sensitive than the mean abundance. In contrast for the short semelparous trait group abundance TOR and mean abundance endpoints reacted in an equal manner to the day of application, whereas the abundance TOA shows nearly no effect at any time of application.

For the long iteroparous (TOR at day 160) and long semelparous (TOR at day 70) trait groups highest effects on abundance at TOR and TOA were observed from day 0 until day 160 with the lowest effects shortly after reproduction. In contrast, for the mean abundance highest effects were found at day 200 and day 71, shortly after the reproduction, for long iteropaorous and semelparous trait groups, respectively.

A different pattern was observed for the short semelparous trait group, for which the highest effects at both endpoints were observed shortly before the reproduction and nearly no effects after the reproduction.

From this analysis it can be concluded that the timing of exposure has an important impact on the effects. The impact can be low or high depending on the species and endpoint. For example a low impact was found for the long semelparous trait group based on mean abundance and a strong impact was found for the short semelparous trait group based on abundance at TOR or the long iteroparous trait group based on abundance at TOA. In general all populations were more susceptible shortly before the reproductive phase. For all species the abundance at TOR as endpoint was the most sensitive.

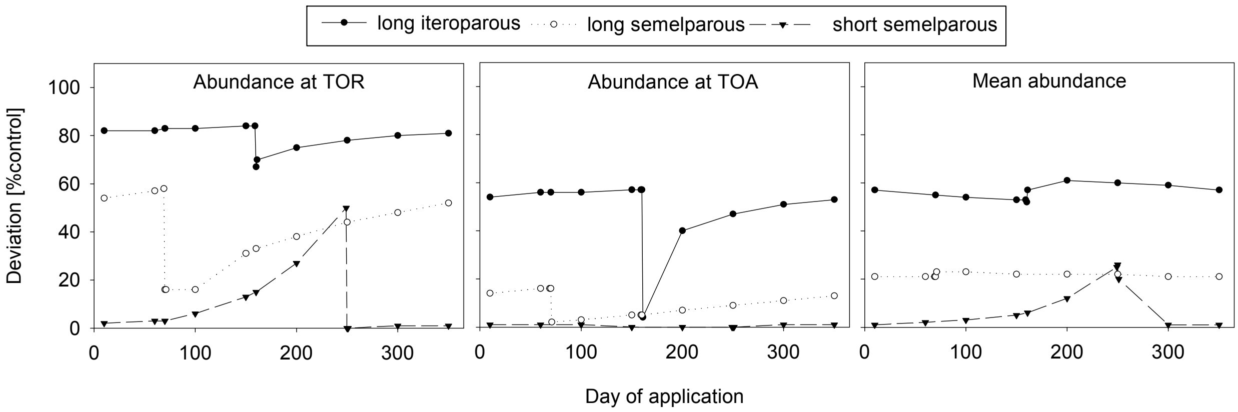

Figure 6-23: Impact of the day of application on the effects on populations after 10 years of yearly exposure.
Three endpoints were calculated, the abundance at the time of reproduction (TOR), the abundance at time of application (TOA) and the mean abundance over the year. The toxicant was applied yearly at a concentration equal to 50 % mortality.

6.2.5.7 Sensitivity analysis

A sensitivity analysis was conducted to investigate the impact of single parameters on the evaluated endpoints: All parameters were increased (up to a factor of 4) and decreased (down to a factor of 10) and the deviation of the endpoints was plotted against the deviation of the parameter (Figure 6-24), this resulting in changes in life-cycle parameters of control population as well as of treated population. The analyses were conducted at one scenario applying a toxicant to get 50 % mortality. As the days of application day 100, 60 and 249 were chosen for the long iteroparous, long and short semelparous trait group, respectively. These days of application result in the highest effects in the populations as demonstrated above. The results clearly indicate that the endpoint abundance at TOR is less sensitive to changes of the life-cycle parameters than the endpoints abundance at TOA or mean abundance. For the endpoint abundance at TOR none of the parameters was sensitive. For the abundance at TOA and mean abundance several parameters influence the calculated effects, in which the parameter offspring number was the most sensitive for the long iteroparous and semelparous trait group. This is important since only for the parameter offspring number appropriate literature data was available and most other parameters were calibrated to result in stable population dynamics.

From this analysis it can be concluded that the parameter offspring number calibrated on literature data is the most sensitive one of the model. Therefore, it can be assumed that the model was calibrated in a realistic manner, even if calibration was based on a small database and for some parameters no data were available.

The endpoint abundance at TOR was not sensitive to changes in life-cycle parameters, since the application was shortly before the reproduction, because this was the most sensitive application window. So the populations were not able to recover from the application within the very narrow time interval between the application and TOR. If recovery is not taken into account life-cycle parameters cannot change the effects on population level and the effects should be equal to the percentage mortality applied.

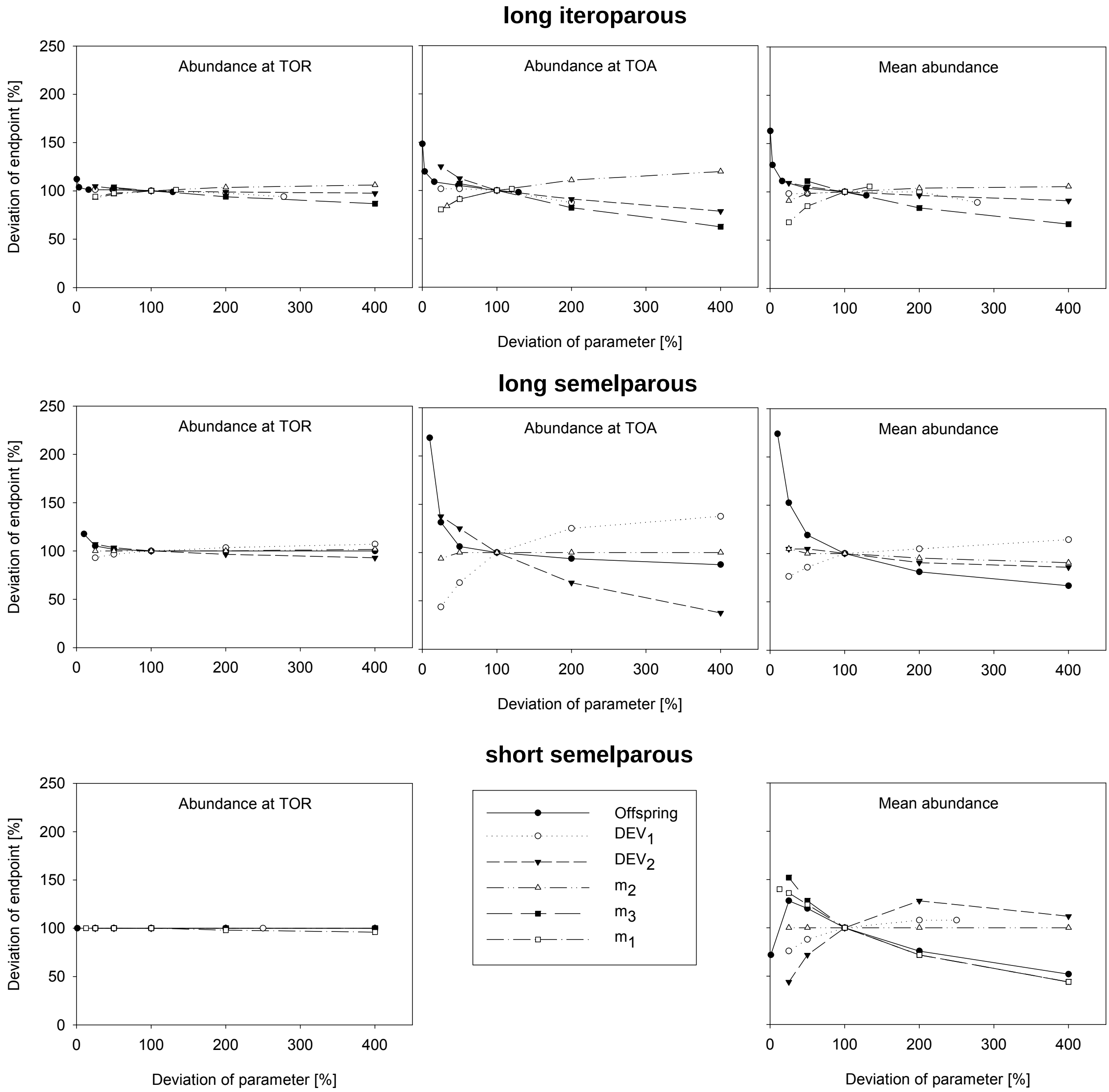

Figure 6-24: Sensitivity analysis of the model parameters.
Three endpoints were calculated, the abundance at the time of reproduction (TOR), the abundance at time of application (TOA) and the mean abundance over the year. The toxicant was applied yearly at a concentration equal to 50 % mortality, the day of application were 100, 60 and 249 for clams, odonates and oligochaetes, respectively. The endpoint abundance at time of applications results always in effects lower than 5 % and is therefore not shown.

6.2.5.8 Application of toxicants to connected populations

6.2.5.8.1 Impact of drift

The evaluation of the importance of drift and the impact of drift parameters to model output were tested on one exposure scenario equal to the worst-case window approach from UBA (2007). Therefore four segments out of 40 were treated with a toxicant equal to 100 % effect and drift parameters as well as two approaches taking drift into account were changed and model output were compared. The two approaches differed in the assumption of the population upstream of the population of interest. This upstream population releases its drift into the first segment of the population of interest. In the first approach (Treatment) it was assumed that the population upstream was in the same state as the population of interest, so damaged by the same amount of toxicant. Therefore the drift from segment 40 was released into the segment 1 of the treated population, so that for this case drift flowed in a continuous circle. The second approach (Control) assumes that the upstream population remains untreated and behaves as a reference. For this approach the control population released its drift from segment 40 into the segment 1 of the treated population. The resulting effect patterns are shown for the long iteroparous trait group in Figure 6-25 and the overall effects on the treated population in the window are shown in Figure 6-26. In the left diagrams of Figure 6-25 the Treatment assumption was used and in the right diagrams the assumption Control.

From the effect patterns in the Treatment scenario it can be concluded that drift results in dispersal of the effect over the population. Without drift only the exposed segments are affected, the number of affected segments increases with increasing drift rates, at a drift rate of 0.005 individuals per day (equal to 0.5 percent drift per day) all 40 segments are affected. At this scenario it becomes obvious that the drift in segment 1 comes from segment 40, resulting in affected segments upstream from the first segment treated with the toxicant. Another important fact of drift is that the effects in the treated segments decrease. It is also remarkably that effects on untreated segments and number of affected segments increase over time and highest effects are found after 10 years. These facts are already known and have lead to the development of metapopulation approaches (e.g. Spromberg et al. 1998). Within a metapopulation approach source and sink populations are defined, whereas the source population (untreated segment) delivers individuals to the sink population (treated segment) resulting in effects on the untreated source population as well as reduced effects on the sink population.

Using the Control scenario a slightly different pattern can be observed. Especially the segments upstream of the first treated segment are not affected. The difference between both scenarios increased with higher drift rates, this reducing the effects on the treated and non treated segments due to immigration from the upstream control population.

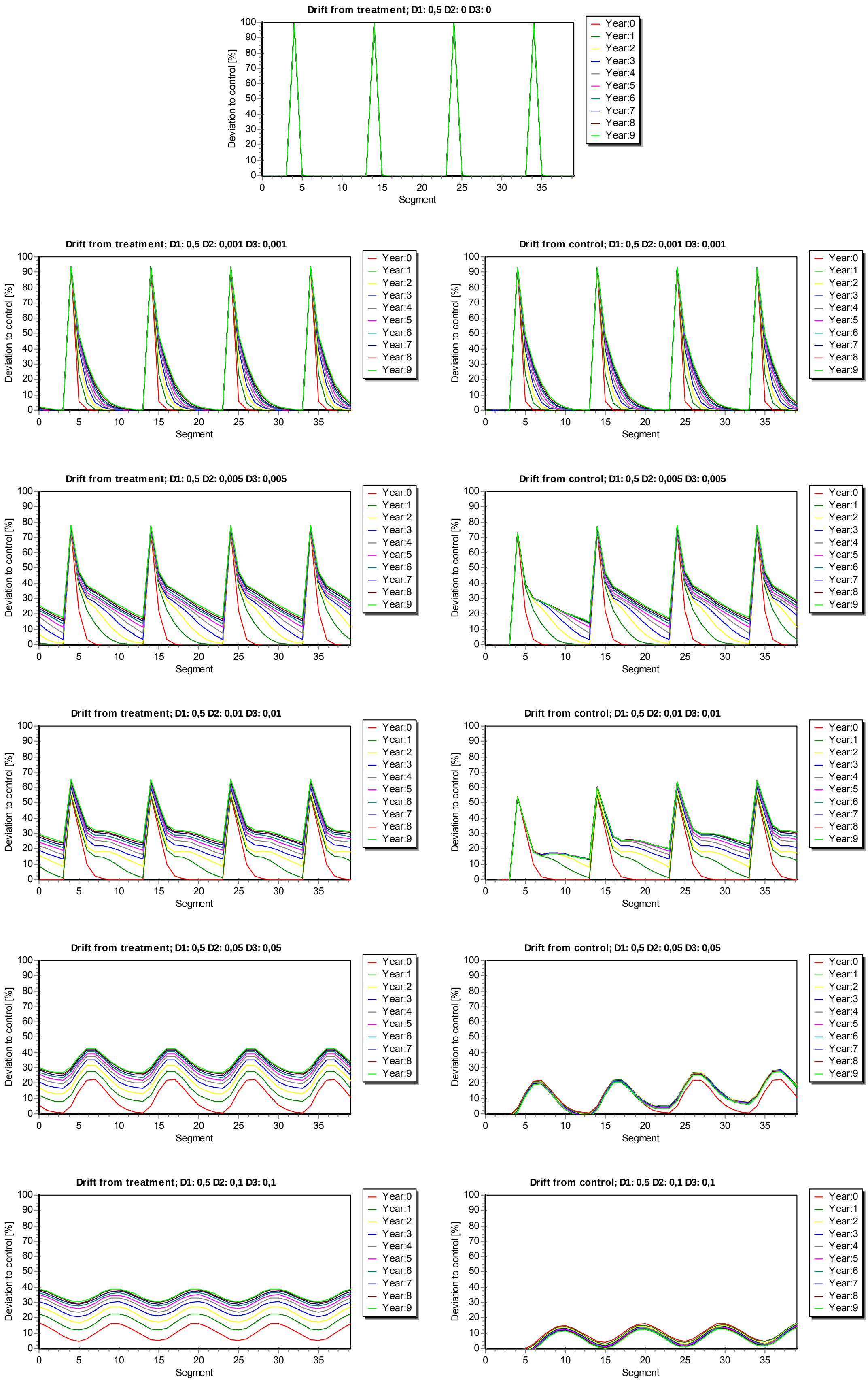

Figure 6-25: Effect pattern on long iteroparous trait group at various drift rates and two scenarios

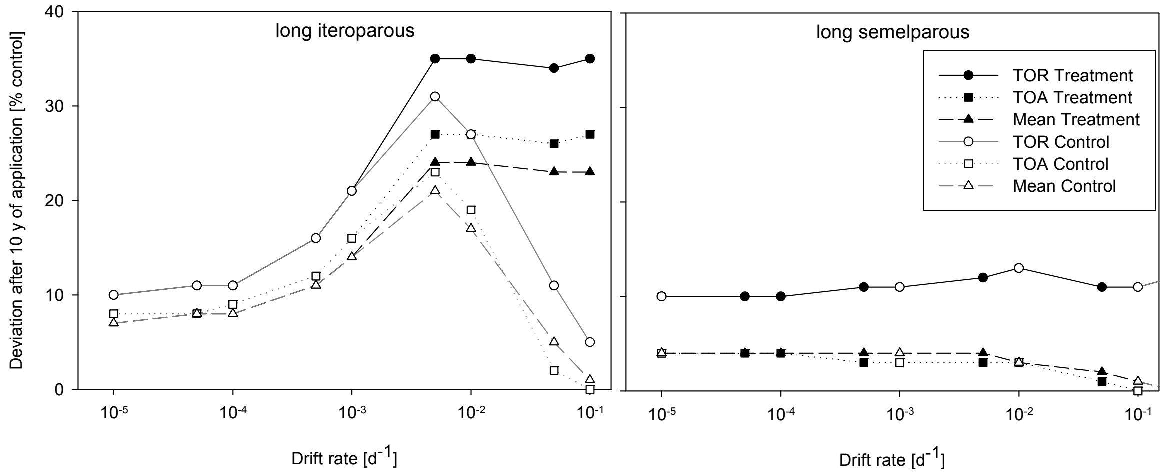

Figure 6-26: Impact of drift rates and scenario assumptions on effects on population.
Two scenarios were simulated, differing in the source population drifting into segment 0 of the population of interest. In the Treatment scenario the source population was assumed to be also treated and in the Control scenario the source population was untreated. Drift rates were equal for stage2 and stage3, whereas for stage1 the drift rate was set constantly to 0.5 individual per day.

6.2.5.8.2 Impact of exposure pattern

For connected populations also the spatial dimension of exposure is important. In the following scenario a population inhabiting 1000 m of a stream (equals to 40 segments) was assumed. Here an effect of 10 % on the population can result from a 10 % effect in each segment or a 100 % effect in 4 segments, each. The impact of spatial exposure pattern on effects on population level after 10 years of yearly exposure was investigated for the long iteroparous trait group at different drift rates, the results are shown in Figure 6-27. With this analysis it was demonstrated that the effects on the population depends on the spatial pattern of exposure. If only a low number of segments is exposed to high concentrations, effects on the overall population are lower than exposure of all segments to a lower concentration. This impact is highest if no drift is assumed and lowest if high drift rate occurs. Therefore, the assumption that effects on populations can be calculated outside of GIS holds not true. Since the spatial exposure pattern has to be taken into account for a realistic calculation it is necessary to conduct the calculation of effects on populations on a geo-referenced base using GIS. This was technically not possible in the current project and thus, a worst case scenario had to be assumed, in which all segments are exposed with the same concentration.

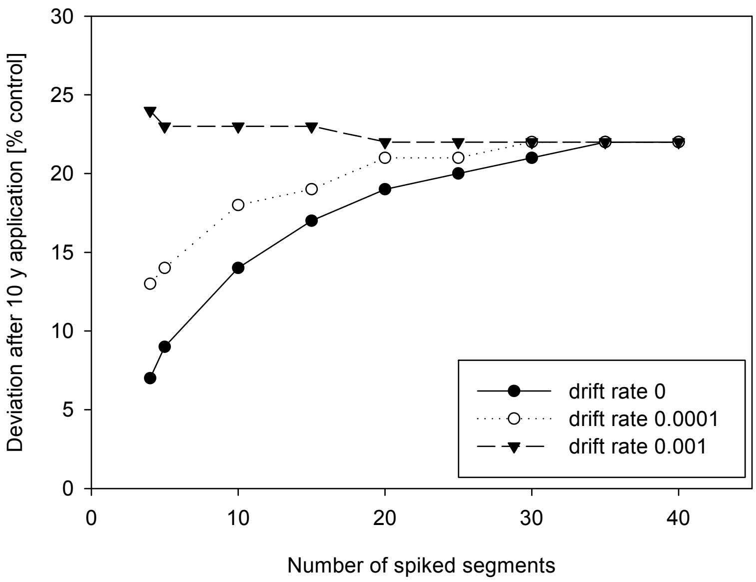

Figure 6-27: Impact of exposure pattern on effects on population of long iteroparous trait group. Different exposure pattern were tested for the effects on the whole population. Therefore different numbers of segments were spiked and the effect per segment was calculated in a way that for each pattern 10 % mortality for the whole population (40 segments) was achieved. As endpoint deviation based on mean abundance is plotted for three different drift rates.

6.2.5.9 Assumptions & limitations of the model approach

The model approach described above has to be seen as a more theoretical effect assessment for a generic approach due to several limitations. First of all, appropriate data for long living species like the investigated species are rare and model calibration was therefore based on a small data base. Since all available data were used for model calibration no model validation was possible. Model validation needs long term population data under the impact of a toxicant. It is not expected that these data will be generate in near future since – as far as we can see - the research funding at the moment allows not such long lasting experiments. Nevertheless the sensitivity analysis indicated that minor changes in single model parameters will not too much influence the calculated effects on the populations. The calculated effects after 10 years of yearly application depend mainly on the life-cycle structure; e.g. for the long iteroparous trait group the long juvenile development and long life span for adults. These life-cycle structures of the investigated species are well known. A more complex modelling approach like individual based models (e.g. Preuss et al. 2009, van den Brink et al. 2006) would allow the implementation of more literature data and a validation on several biological levels and thus reduce uncertainty of the model approach. We believe that a meaningful individual based model might be developed for the investigated species, but this is out of scope of the current project.

Several ecological relevant mechanisms were not implemented in the modelling approach. These include those which are increasing recovery of populations and thus increasing population sustainability, such as interaction with the food resource, recolonisation as well as different sensitivity of various life-stages. Not including these mechanisms makes the approach conservative. Also mechanisms reducing population sustainability were not included making the approach less protective. These mechanisms include competition, predation and indirect effects. The uncertainty of disregarding these mechanisms in the modelling approach cannot be quantified yet.

6.2.5.10 Derivation of tolerable effects

To derive the tolerable effects simulations were performed over 10 years of yearly application. For the factors influencing recovery of populations mentioned in the above sections the worst case approach was used for the generic analysis.

In general all populations proved to be more susceptible shortly before the reproductive phase. In order to consider this, different application patterns to calculate the effects on populations were selected for the three trait groups. In the long iteroparous trait group TOA was day 100, for the long semelparous one day 60 and for the short semelparous one day 240. Since the susceptibility of a population is markedly dependent on the drift and exposure patterns for the time being no solid basis is available to selecting the appropriate scenario which – as a further challenge - has to be spatial explicit, tolerable effects on single populations were calculated as described in 6.2.5.8.2. There the scenario was demonstrated as the worst case approach. The effects on the populations for different levels of mortality are given in Table 6-6.

For all three representative species analysed here the abundance at TOR as endpoint was more sensitive than the abundance at TOA and the mean abundance; it is also minor sensitive to changes in life-cycle parameters. But there is one disadvantage in this endpoint: it does not take recovery into account if application is shortly before the TOR as was calculated in this approach. This becomes obvious from Table 6-6, in which the relationship between mortality and effects show a linear dependence for short and long semelparous trait groups. Only for the long iteroparous trait group this endpoint shows a non linear relationship. In contrast, both the abundance at TOA and the mean abundance of all three species show non-linear relations, because these endpoints are not only related on the direct effect but also on recovery within one year. Therefore, **the tolerable effects were derived from the endpoint TOA for the long semelparous and iteroparous trait groups.** The endpoint mean abundance was used for the short semelparous trait group. **It was assumed that 20 % effect after 10 years of yearly application does not result in dramatic effects on the population level**. In most standardized biotests the minimum detectable difference is around 20 % or higher, i.e. the calculated NOEC equals a 20 % effect (Pohl et al. 2006).

In doing so, the tolerable effect levels for the three species proved to be 50, 10 and 30 % mortality for long semelparous, long iteroparous and short semelparous trait groups, respectively.

Table 6-6: Effects on population abundance at different levels of mortality

Mortality due to toxicant [%]	***Caloptery*** **Long semelparous**			***Anodonta*** **Long iteroparous**			***Dero*** **Short semelparous**		
	TOR	TOA	Mean	TOR	TOA	Mean	TOR	TOA	Mean
0	0	0	0	0	0	0	0	0	0
5	6	1	2	13	9	7	5	0	3
10	12	3	3	25	17	14	10	0	6
20	24	5	7	45	32	27	20	0	13
30	35	8	11	60	43	36	30	1	20
40	47	12	15	71	51	43	40	1	27
50	57	16	21	79	57	48	50	1	35
60	68	20	27	85	62	53	60	2	43
70	77	25	35	90	66	57	70	2	52
80	86	32	45	94	69	61	80	3	62
90	94	44	62	97	74	67	90	5	73
100	100	100	100	100	100	100	100	100	100

A more detailed analysis of the effect levels for the three trait groups was conducted and the results are shown in the following diagrams. The endpoints abundance at TOA and TOR as well as the mean abundance over time at the effect thresholds are shown in Figure 6-28 together with the deviation from control in Figure 6-29. From both figures it can be concluded that all effects on all three endpoints increases within the first years followed by no further increase within the subsequent years. Therefore, it can be concluded that even longer time intervals of applications will not increase the effects on populations. Figure 6-29 clearly demonstrates that all populations reach the control level in some parts of the year, except the long semelparous trait group in which only 95 % of the control level was reached.

Thus, it is assumed that at the determined effect levels the populations are affected to a given extent, but that these effects are so small, that they could not be detected in (semi-) field studies (e.g mesocosm studies or field monitoring) and will not lead to extinction of the populations.

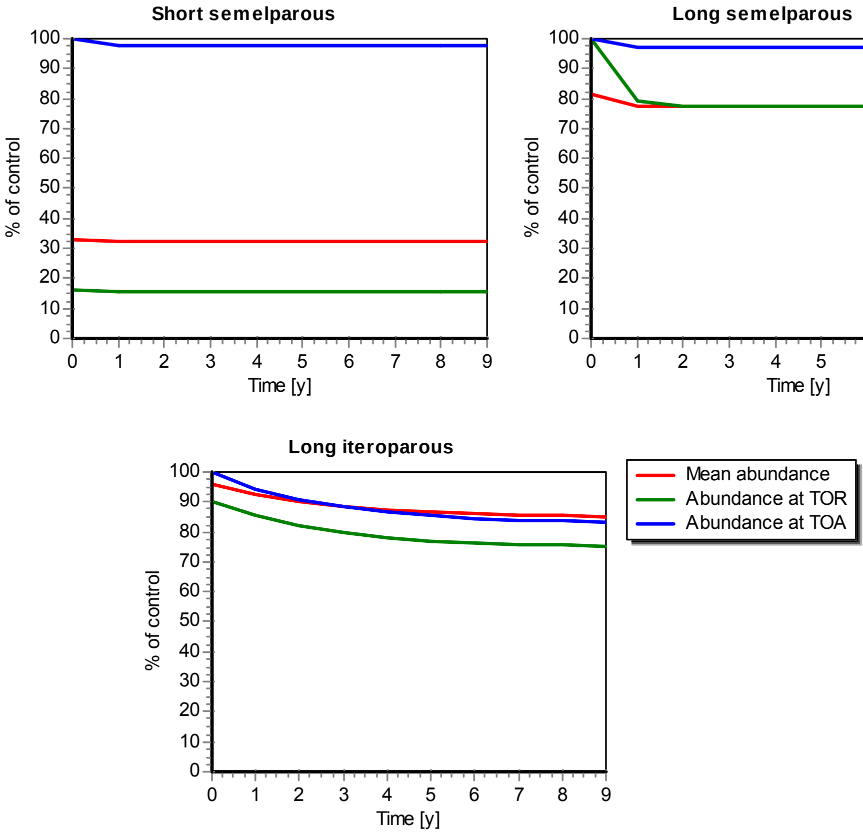

Figure 6-28: Calculated endpoints resulting from application of a theoretical toxicant at the effect threshold for the three trait groups over ten years.
Three endpoints were calculated, the abundance at the time of reproduction (TOR), the abundance at time of application (TOA) and the mean abundance over the year. The toxicant was applied yearly at the effect threshold, namely 30, 50, 10 % mortality at day 240, 60, 100 for the short semelparous, long semelparous and long iteroparous trait group respectively.

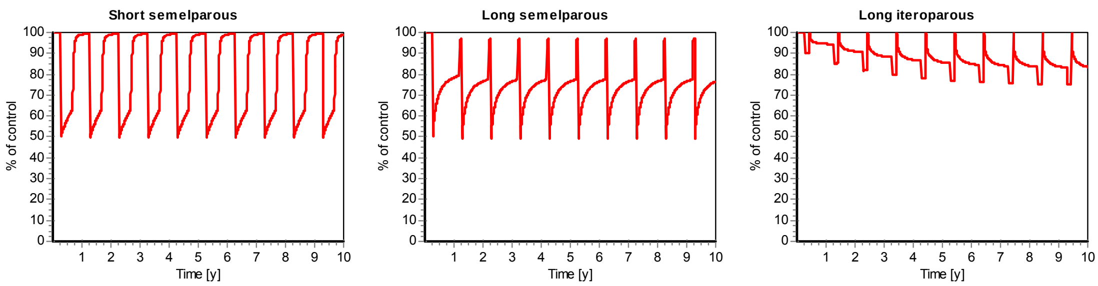

Figure 6-29: Effects resulting from application of a theoretical toxicant at the effect threshold for the three trait groups over ten years.
The toxicant was applied yearly at the effect threshold, namely 30, 50, 10 % mortality at day 240, 60, 100 for the short semelparous, long semelparous and long iteroparous trait group respectively.

The calculation of the tolerable effects was based on worst case assumptions about the spatial pattern of exposure as well as about the drift of organisms. To highlight the potential of more realistic simulations, the most sensitive species (representing the long iteroparous trait group) was investigated under more realistic conditions simulation the tolerable effect. In doing so, it was assumed that only 4 out of 40 segments were exposed, resulting in 100 % mortality each, and that realistic drift rates are comparably low (70 % per day for stage 1, 0.02 % per day for stage 2 and 0.01 % per day for stage 3). The calculated effects after 10 years of yearly exposure were 11, 8 and 8 % for abundance at TOA, at TOR and mean abundance respectively. So, the calculated effects under this more realistic scenario equal about half the effects calculated in the worst case scenario (for detailed results see Figure 6-30, Figure 6-31 and Figure 6-32). These figures demonstrate that at the applied spatial exposure pattern in four segments dramatic effects can be observed but the overall population is only slightly affected. Therefore, the conclusion appears to be justified that with a geo-referenced calculation the effects on populations are probably lower by a factor of around 2. In doing so, the developed population model has to be implemented in GIS, which needs a high amount of computer capacity. However, this is possible today and together with a parallelisation modelling approach geo-referenced population effects would be computable, but was out of scope for the current project.

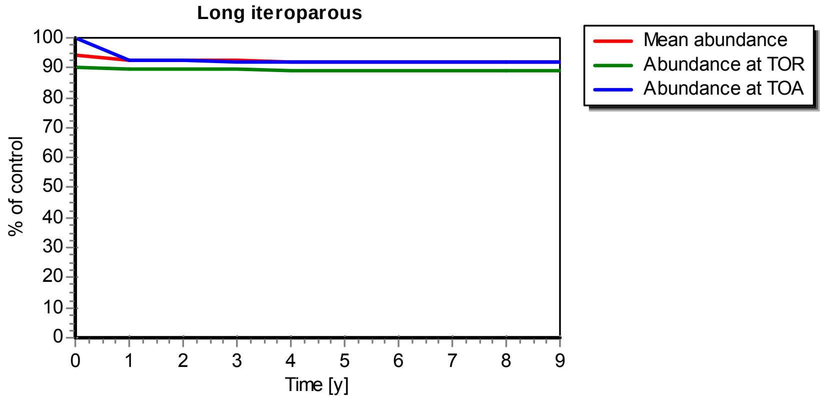

Figure 6-30: Calculated endpoints resulting from application of a theoretical toxicant at the effect threshold for the long iteroparous trait groups over ten years using a more realistic scenario. Three endpoints were calculated, the abundance at the time of reproduction (TOR), the abundance at time of application (TOA) and the mean abundance over the year. The toxicant was applied yearly at the effect threshold, 10 % mortality at day 100.

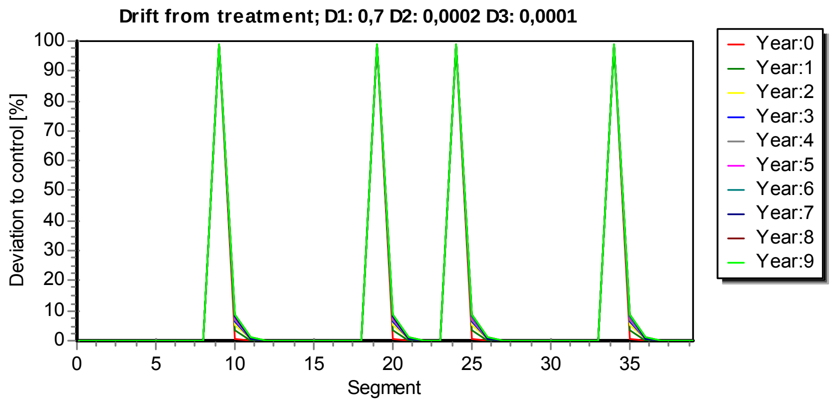

Figure 6-31: Effect pattern on long iteroparous trait group

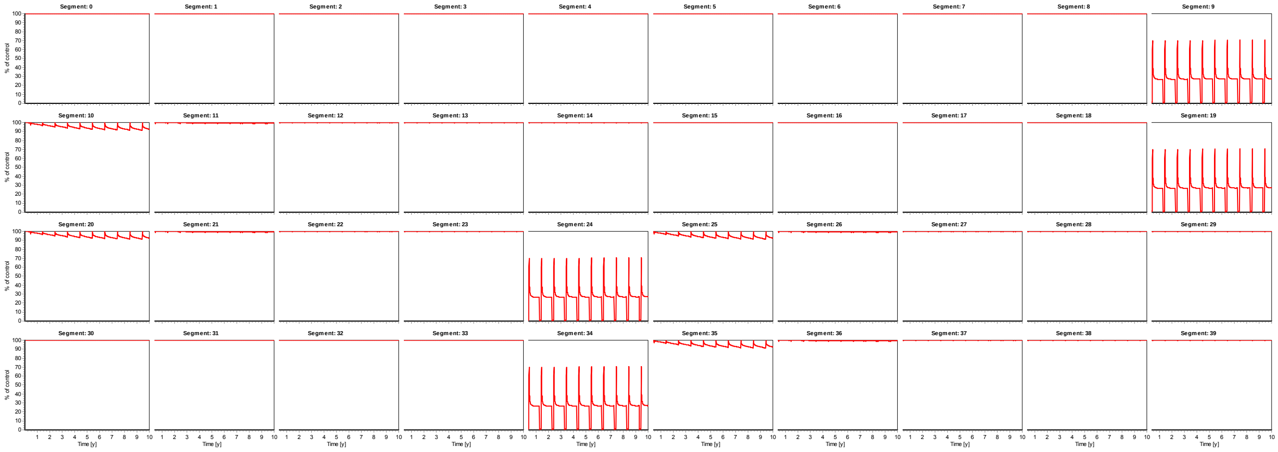

Figure 6-32: Effect pattern on long iteroparous trait group in each segment

6.2.5.11 Conclusions

Using a modelling approach taking the life-cycle of realistic worst-case species into account it was demonstrated that effects on populations due to yearly application can increase over time, which strongly indicates that the effects depend on the life-cycle characteristics. Strongest effects were found in species (long iteroparous) with a long life span and a juvenile development lasting more than 2 years, whereas for univoltine species (short semelparous) recovery within one year was demonstrated. Sensitivity analysis of the model indicates that calculated effects on population as model output proved to be robust for the assumptions made for model calibration.

Additionally it was demonstrated that timing of application and spatial exposure patterns influence the overall effects on population in the field. For both of these factors worst case assumptions were used to calculate the tolerable effects on population level. The results from this analysis indicate that **10 % mortality for the population** of the long iteroparous trait group **is acceptable and should therefore be used in the GeoRisk-Project for the generic approach**.

For substance related assessment the information on the sensitivity of different taxa should bes used and therefore refineda hotspot criteria should be derived and applied.

As a first approach, the following tolerable effects levels (once a year) are suggested:

- Phyto- and zooplankton species: 90 %
- Invertebrates with short semelparous life cycle but low dispersal (e.g. *Dero digitata*): 30 %
- Invertebrates with long semelparous life cycle and medium dispersal (e.g. *Calopteryx virgo*): 50 %
- Invertebrates with very long iteroparous lifecycle and low dispersal (e.g. *Anodonta cygnea*): 10 %
- For vertebrates and macrophytes: 10 %

Nevertheless future approaches in which the effects are calculated in a spatially explicit way by implementing the population model in GIS can lead to refined calculations resulting even in higher tolerable effects for the populations of interest. More advanced modelling techniques, e.g. individual-based population modelling, were out of scope for the current project but would allow implementation of further knowledge about the species in future. Due to better possibility for model testing and stochastic model nature this kind of models can decrease model uncertainty and increase model acceptance.

6.3 Calculation of the magnitude of effects in risk segments

6.3.1 Estimation of a slope for the dose-response relationships in the generic assessment

In the first proposal for the application of hotspot criteria by the UBA (2007) a 100 % (lethal) effect is assumed on a population in a water body segment if the PEC exceeds the RAC. This is clearly overprotective considering that the RAC is as kind of threshold concentration and it is comparable to the PNEC (Predicted _No Effect Concentration) used in the risk assessment for other chemicals than plant protection products. For example, for fish the RAC is equal to the lowest acute LC_{50} divided by a factor of 100 or the lowest chronic NOEC divided by 10. Thus, at least if the RAC is only slightly exceeded, a 100 % effect is unlikely.

It was intended to derive slope of dose response relationships of worst case substances provided by the UBA. However, these datasets could not be provided. Alternatively the US EPA Ecotox data base was checked but it was found to be not possible to extract datasets for calculation of slopes within this project.

Therefore, available data for carbaryl were used as a first estimation of a slope for a realistic worst case substance. Carbaryl acts by inhibition of the acytylcholinesterase. Dose-response functions for different macroinvertebrates are very similar with slopes of 3.9 – 4.6 with a mean of 4.3 (Figure 6-33).

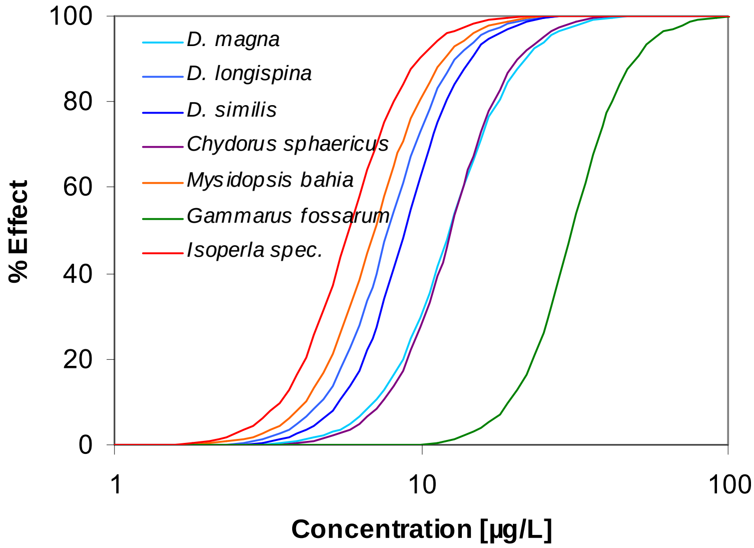

Figure 6-33: Dose-response curves for acute effects of carbaryl on different aquatic invertebrates (data from C. Schäfers, not published)

A comparison of the logistic dose-response function with different slopes is shown in the following diagram. For example, for a slope of 4 the EC_{10} and EC_{50} differ approximately by a factor of 10.

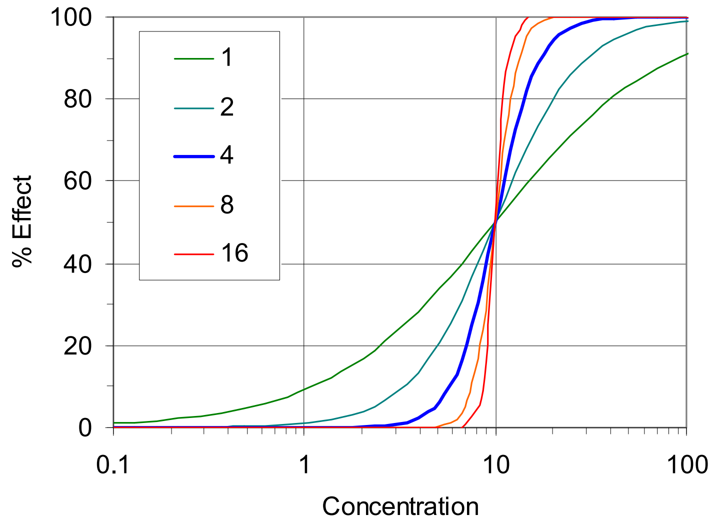

Figure 6-34: Logistic dose response curve for an EC_{50} of 10 µg/L and different slopes

The following approach is suggested to consider dose-response relations in generic and product specific assessments:

- As a first conservative step in a **generic assessment**, 100 % lethal effect for PEC > RAC could be used.
- If magnitude of exposure is indicated as multiple of the RAC it could easily be decided if a refinement of the slope would change the result (i.e. the number of management segments): If PEC are in most cases higher then ten times the RAC, the use of a dose-response function will usually not change the result significantly.
- If a considerable number of risk segments are characterized by PEC < 10 x RAC the analysis should be refined assuming a realistic worst case slope, e.g. 4 derived for carbaryl. If other worst case substance data are available, this could be refined.
- To calculate the effect per segment from the PEC, it is suggested to use as the RAC the EC_{10} of the dose response relation: The RAC is assumed as 'safe' concentration, thus not leading to an unacceptable ecological effect. A 10 % effect is considered here to be acceptable because NOECs and EC_{10} are often considered to be exchangeable in the use in the risk assessment and 10 % mortality in the controls is often accepted for the validity of the test.
- For a **product related assessment**, the slope of the dose response curve for the most sensitive taxon should be used if available and – as in the generic assessment - the RAC should be used as the EC_{10}. For example, in the standard acute risk assessment for fish, the RAC would be the lower of two LC_{50} divided by 100 and a dose-response function with an LC_{10} = RAC and the slope of the dose-response of the most sensitive fish should be protective for other fish species, too. The similar approach could be used for EC_{50} of invertebrates as well as algae and macrophytes (standard trigger 10).
- For long-term tests, e.g. the *Daphnia* reproduction test or fish juvenile growth test, the RAC is based on the NOEC divided by a factor of 10, but it could also be based on the EC_{10}. However, within the project the focus is on acute effects, respectively also for an

RAC derived from chronic studies, conservatively lethal effects assumed in the hotspot analysis.

- If the RAC is based on an SSD the approach would be the same (only the RAC is derived by the use of a smaller safety factor). With respect to the slope to be used, the steepest slope of the species tested in the SSD should be used for a worst case estimation.
- Micro- and mesocosm studies are used to refine the risk assessment for algae, macrophytes and/or invertebrates. The NOEC for the different groups can be used as NOECs from a laboratory test but with a reduced safety factor (as derived in the usually risk assessment based on such a study). Thus, divided by the safety factor it could serve as the EC_{10} for the dose response curve for the hotspot identification. However, the NOEAEC of a micro- or mesocosm study is based on the recovery of the affected populations. Because recovery is included in the derivation of the hotspot criteria for the different groups, the NOEAEC should not be used to derive a RAC for hotspot identification. However, if the micro- or mesocosm study demonstrates faster recovery than considered in the hotspot criteria, refinement might be discussed on a case by case basis.

6.3.2 Consideration of short-term exposure

The GeoRisk approach is based on realistic hydrodynamic data and a dynamic exposure model for running waters. Monitoring data revealed that the width depth ratio of streams is often higher than the 3.3 assumed in the current standard scenario. Therefore, initial PEC values for small water bodies are often higher than those obtained with the standard static model assuming 30 cm water depth for a ditch of 1 m width. However, local exposure duration can be significantly shorter than in static waters and in the ecotoxicological tests. Usually, e.g. for the FOCUS stream scenarios, the risk assessment is conducted in a conservative manner by comparing the Regulatory Acceptable Concentration (RAC, e.g. the LC_{50} obtained in an acute test over 96 h of exposure) to the maximum Predicted Environmental Concentration (PEC). However, this can clearly be overprotective for very short exposure events which can occur in running waters.

6.3.2.1 Aim

While most ecotoxicological tests try to keep the exposure concentration constant, exposure profiles for plant protection products in the field can often be expected to be very variable over time, e.g. due to different entry routes, fast dissipation or transport and dispersion in running waters. The aim of this chapter was to identify a realistic worst-case exposure time to effect relationship, which easily allows the integration of RAC depending on time of exposure for the hotspot analysis.

6.3.2.2 Theoretical background

Within a standard toxicity test, the species of interest is incubated to a constant exposure of the test item for a certain time at which a concentration-response relationship is determined, but toxicity is a dynamic process. It is a general fact that for most toxicants the LC_{50} decreases over time. In other words, a higher concentration is needed to evoke 50 % mortality after 1 h than after 96 h. At a given time point equilibrium of toxicity is reached and effects will not increase further on (McCarty & Mackay 1993). This time point depends on the species considered as well as on the mode of action and the physico-chemical properties of the test item. This dynamics of toxicity is not taken into account at standard toxicity tests nor in the standard risk assessment.

Modelling the fate of the compounds more realistically asks to answer the question about what effects the resulting exposure pattern will have on the population level in aquatic organisms. The uncertainty in predicting effects from time-variable exposure is discussed in detail by Reinert et al. 2002.

Since toxicity is a dynamic process, which leads to decreasing survival probabilities over exposure time, various empirical models taking this dynamic into account were developed (Ashauer et al. 2006, Sanchez-Bayo 2009). All these empirical models take - beside the concentration - also the exposure time into account. Other mechanistic effect models take the toxicokinetics and toxicodynamics into account. These effect models were recently compared by their underlying hypothesis and mathematical constrains (Ashauer & Brown 2008) and it was shown that the damage assessment model is the most generalized form from which the other effect models can be derived using some different assumptions of the toxicokinetics and toxicodynamics. The underlying principle of the damage assessment model (DAM) is that the internal concentration at the target site triggers the effect and not the external exposure concentration (McCarty & Mackay 1993) and that the effect depends on exposure time. Therefore all forms of DAM (Ashauer et al. 2007a, Lee et al. 2002a) assume first of all toxicokinetics, describing the uptake and excretion of the test substance within the organism. Coupled to this toxicokinetic model is a first order toxicodynamic model. With these kinetic models it seems possible to extrapolate between different exposure scenarios as well as to compare different substances and different species in a more mechanistic point of view.

All these model approaches are currently not applied in ERA and need further data or at least further data analysis and their use was therefore out of scope within this project and a more simple approach was choosen.

6.3.2.3 General approach

The basic idea is to identify realistic worst case substances with respect to the speed of their toxic effect (the faster the effects the less important is the duration of the exposure) and to fit a relationship for the LC_{50} depending on exposure duration related to the duration of the standard test. If this relation is fitted to a realistic worst case substance it should be protective for other substances with a slower mode of action as well.

Therefore a literature survey was conducted to find publications which describe (acute) toxicity of freshwater organisms (invertebrates and fish) depending on duration of exposure.

For algae and macrophytes the primary endpoint of the standard tests is inhibition of growth and under EU 91/414 these tests are considered as chronic tests. For these tests, as well as for other chronic tests (e.g. *Daphnia* reproduction test, fish early life stage or juvenile growth test) the approach outlined in the elLink recommendations (Brock et al. 2009) should be considered, i.e. it should be checked if and for which time window the time weighted average approach could be used.

Thus, the focus here is on the extrapolation of a RAC derived from the acute *Daphnia* test, other invertebrate or fish tests with 48 or 96 h duration to a RAC applicable to situations with significantly shorter exposure.

The datasets were organized in a data base (in Microsoft Excel) and categorized according to their reliability and relevance to derive such relationship:

Table 6-7: Classification of datasets in the literature review on effects of pulse exposure

Group	Description	Use for extrapolation to shorter exposure
1	Different exposure durations at the same test durations	Directly usable
2	Same exposure durations at different test durations with a focus on additional effects after exposure	Seperate issue, latency of effects
3	Different exposure durations without the same observation time	Usable with limitation, that latency of effect until the standard test duration is unknown.
4	Unspecified observation time	With limitation
5	Data not in the literature. But there are for-mula and parameters to identify the data probably.	To be checked
6	Only one exposure duration	Not useful, because of incomplete information or only one exposure duration tested

From the reliable and relevant datasets three types of functions were addressed from the da-tasets, which were compared and a realistic worst case relationship to estimate effects at short exposure duration was selected. The three different functions were determined on current risk assessment practice, differing for daphnia and other invertebrates as well as for fish:

- *RAC based on the standard acute Daphnia test (exposure over 48 h)*
- *RAC based on several acute invertebrate tests (exposure over 96 h)*
- *RAC based on acute fish tests (exposure over 96 h)*

6.3.2.4 Literature research and data base

Finding time dependent LC_{50} in literature is quite complex, since normally neither the title nor the keywords indicate such kind of data if not the whole manuscript deals with toxicodynamics. Therefore, literature dealing with toxicodynamics was evaluated as well as literature known by the authors of this chapter. Additional the Ecotox database of the US-EPA (http://cfpub.epa.gov/ecotox/) was screened for studies with several timepoint and especially with exposure periods below 12h. From this analysis 511 datapoints were collected from 41 manuscripts for 104 substances and 53 species. The categorisation of the datasets was as fol-lows:

Table 6-8: References related to the different categories of data on time dependent effects

Group	**Literature**
1	Jarvinten et al. 1988
	Kreutzweiser et al. 1994
	Maund
	Peterson et al. 2001
	Schäfers 2002

	Schuller et al. 2006
2	Beketov & Liess 2007
	Van der Hoeven & Gerritsen 1997
3	Adema 1978
	Andersen et al 2006
	Armstrong 1976
	Bailey et al. 1999
	Brinkmann & Kuhn 1977
	Buhl et al. 1993
	Grushko et al. 1975
	Hunt 1981
	Kopperman et al. 1974
	Kreutzweiser et al. 1994
	LeBlanc & Surprenant 1983
	Legierse 1998
	Pickering & Henderson 1966
	Rubach et al 2010
	Sanders & Walsh 1975
	Schäfers 2002
	Ural & Saglam 2005
	Van der Hoeven & Gerritsen 1997
	Van Heerden et al. 1995
4	Hermens, Unpublished
5	Heming et al. 1989
6	Barron & Woodburn 1995
	Beketov 2004
	Faust et al. 2003
	Forget et al. 1998
	Holdway et al. 1994
	Ibrahim et al. 1998
	Könemann 1981
	Kreutzweiser et al. 1994
	Lukancic et al. 2009
	Palmquist et al. 2008
	Pamela et al. 1997
	Richards & Baker 1993
	Solomon et al. 1996
	Tomlin 1997
	Wacksman et al. 2006
	Walter et al. 2002
	Wang et al. 2004

6.3.2.5 RAC based on the standard Daphnia EC_{50} of 48 h

No data were found for exposure of *Daphnia spec.* less than 24 h. However, 25 data pairs of EC_{50} for 24 and 48 h were given, which allow estimating a factor to extrapolate from 48 h- EC_{50} to 24 h-EC_{50}.

Most of the datasets were obtained with *D. magna*, one with *D. longispina* and three with *D. pulex*. For one substance (chlorpyrifos) data pairs were found for two different species.

The 24h-EC_{50} was 1.0 to 14.4 times higher than the 48h-EC_{50}. The median was 1.6, the 10. percentile was 1.2.

Table 6-9: Ratios of 24h-EC_{50} to 48 h-EC_{50} for *Daphnia spec*.

Substance	24h-LC_{50} / 48h-LC_{50}	Stage	Conditions
1,1,2-trichloroethane (TCE)	1.00	1 day old	fed
1,1,2-trichloroethane (TCE)	1.02	1 day old	not fed
Acetone	1.13		
Primicarb	1.25	Neonates	
Carbaryl	1.27		
3,4-dichloroaniline (DCAn),	1.31	7 days olds	not fed
Dimethyl formamide	1.31		
3,4-dichloroaniline (DCAn)	1.33	7 days olds	fed
Chlorpyrifos (Dursban 4E)	1.38	Juvenils	
Dimethoate	1.45	Neonates	
m-nitrophenol	1.63		
pentachlorophenol (PCP)	1.63	7 days olds	not fed
1,1,2-trichloroethane (TCE)	1.63	7 days olds	not fed
Triethylene glycol	1.66		
Chlorpyrifos (technical-grade)	1.68	Neonates	
pentachlorophenol (PCP)	1.70	1 day old	fed
3,4-dichloroaniline (DCAn)	1.72	1 day old	not fed
1,1,2-trichloroethane (TCE)	1.74	7 days olds	fed
pentachlorophenol (PCP)	1.87	7 days olds	fed
3,4-dichloroaniline (DCAn)	2.00	1 day old	fed
pentachlorophenol (PCP)	2.00	1 day old	not fed
Chlorpyrifos (Dursban 4E)	2.00	Neonates	
Phenol	3.23		

p-nitrophenol	4.17		
Chlorpyrifos	14.40		

In a few studies, effects of exposure longer than 48 h or latency of effects were analysed with *Daphnia.* These studies are discussed within the section on aquatic invertebrates or on latency of effects.

Suggestion for application in a geo-referenced risk assessment:

- Check if predicted exposure duration is significantly shorter than in the test (consider measured concentrations in the test)
- If the RAC is based on the acute *Daphnia* EC_{50} and the PEC is below the RAC = $EC_{50/100}$ less than 24 h, the substance specific EC_{50} for 24 h should be used if available. Because uncertainty regarding latency of effects is not changed compared to the standard test, the trigger value of 100 should be applied.
- If only the 48h EC_{50} is available, a more realistic but still conservative estimation of a RAC for short exposure (< 24 h) could be done by multiplying the standard EC_{50} by 1.6 (the median of the data analysed here) and applying the standard trigger value.

6.3.2.6 Pulse RAC based on 96 h EC/LC$_{50}$ for invertebrates

For the GeoRisk workshop in November 2009, carbaryl was used as a realistic worst case example because due to its mode of action (inhibition of acetylcholinesterase) lethal effects on invertebrates are expected to be very fast. In addition to LC_{50} values for exposure over 96 h data were also available for 1 hour pulse exposure (followed by 95 h in untreated medium) for three sensitive insect taxa which indicate that an LC_{50} for 1 h exposure can be expected to be at least by a factor of 5 higher than for 96 h exposure. To obtain a relationship of LC_{50} from exposure duration, the EC_{50} over 24 and 48 h from another sensitive species, *D. longispina*; were also considered.

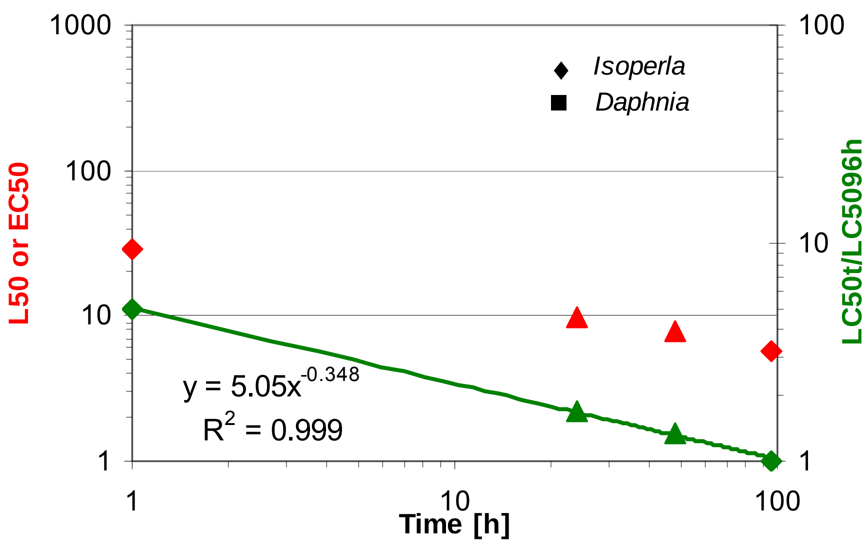

Figure 6-35: LC_{50} depending on the duration of exposure based on carbaryl data for *Isoperla spec.* and *Daphnia longispina*

Examples for a good fit of a power function have been found for several combinations of test substance and test species:

Beketov & Liess (2008) report the time to reach 50 % mortality (ET_{50}) for three species exposed to Thiacloprid (Figure 6-38). Sanchez-Bayo (2009) determined ET_{50} for *D. magna* and *Cypridopsis spec*. for several concentrations of Imidacloprid. For the highest test concentration the ET_{50} was 72 h, thus the data do not allow extrapolation to shorter exposure as tested in the standard test. However, the data showed also a reasonable fit to a power function (Figure 6-36).

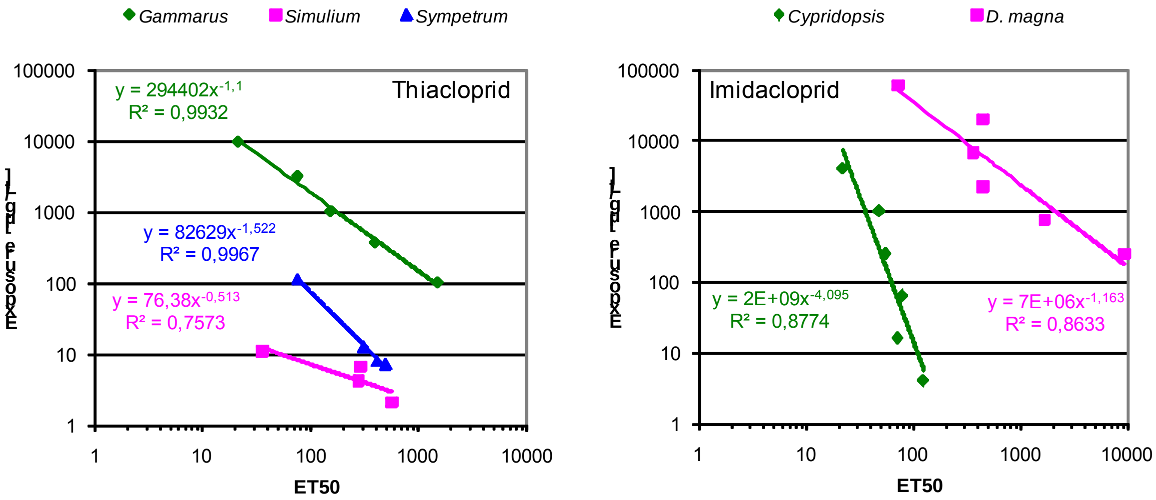

Figure 6-36: ET_{50} depending on the exposure level for several species tested with Thiacloprid or Imidacloprid

LC_{50} values for different exposure durations are reported for *Lymnae stagnalis* and Chlorthion (Legierse 1998), *Hyalella azteca* fo Cypermethrin (Maund et al. 2001) and *Cancer magister* for Methodychlor (Armstrong et al. 1976) and are well fitted by a power function (Figure 6-37).

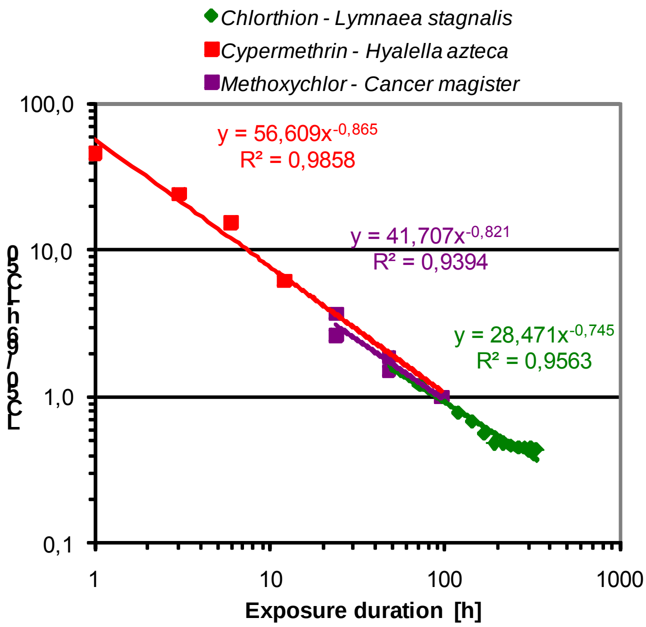

Figure 6-37: Relative LC_{50} depending on the duration of exposure for various compounds and invertebrate species (references see text)

Rubach et al. (2010) measured immobility of *D. magna* after 24, 48 72, and 96 hours of exposure. Within 48 and 96 h, the log(EC_{50}) was inversely proportional to the log of the exposure duration. Below 487 hours, the EC_{50} increased more strongly.

Van der Hoeven & Gerritsen (1997) conducted several tests with *D. pulex* and chlorpyrifos with different exposure durations and effect observation times. Toxitcity increased less with exposure duration, respectively, compared to the results of Rubach et al. (submitted) for *D. magna* who found an EC_{50} for 24 h 14 times larger than the EC_{50} for 48 h while the factor was around 1.5 for *D. pulex*.

In both cases, a power function could be fitted with a R^2 above 0.9 (Figure 6-38).

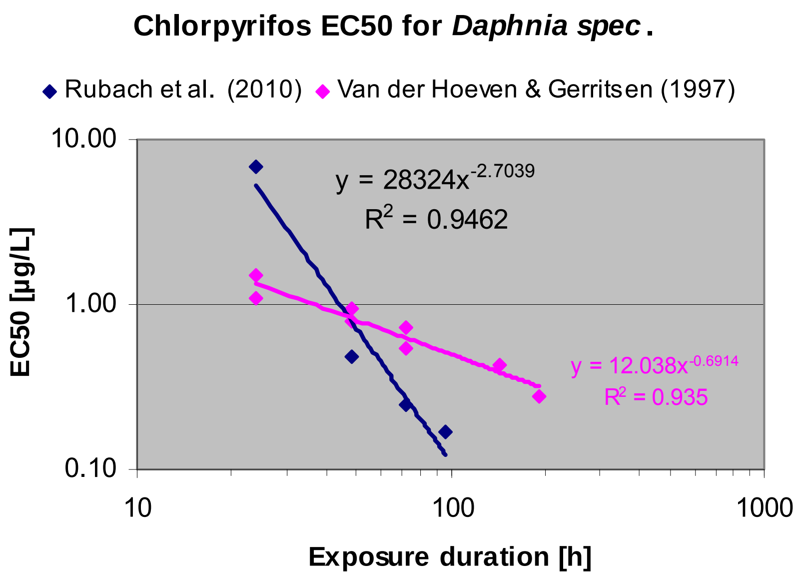

Figure 6-38: EC_{50} depending on exposure duration for effects of chlorpyrifos on *Daphnia spec*.

Overall it can be concluded that for invertebrates a power function is able to describe the changes of LC_{50} over time, even for very short term exposure. The slope of the power functions were ranging from 0.347 to 4. The most shallow regression were found for carbaryl acting on *Isoperla spec*. with a slope of 0.347 which can be used as a realistic worst case relationship to calculate RACs depending on exposure time for aquatic invertebrates.

6.3.2.7 Pulse RAC based on 96 h LC_{50} for fish

The data base includes four date sets where the effects of different exposure durations were tested and survival over the standard acute test duration of 96 h was monitored. Thus, latency of short-term effects until 96 h could be observed.

Jarvinten et al. 1988 (cited in Handy 1994) analysed the effects of shortened exposure duration on the 96 h survival rate of the fathead minnow (*Pimephales promelas*) for chlorpyrifos, endrin, and fenvalerate. The 96 h-LC_{50} for the different exposure durations were related to the standard LC_{50} (exposure over 96 h), and a power function was fitted to estimate the LC_{50} from exposure duration. The first LC_{50} given originally as > x was conservatively used an x for the fitting.

The data and the resulting fits are shown in Figure 6-39 (left figure). For estimating LC_{50} for exposure below 96 h the function for chlorpyrifos is protective also for endrin and fenvalerate and also for the effects of trichlopyr ester, tested by Kreutzweiser et al. 1994 with the rainbow trout (Figure 6-39, right): Chlorpyrifos has the smallest slope of the four chemicals (0.5).

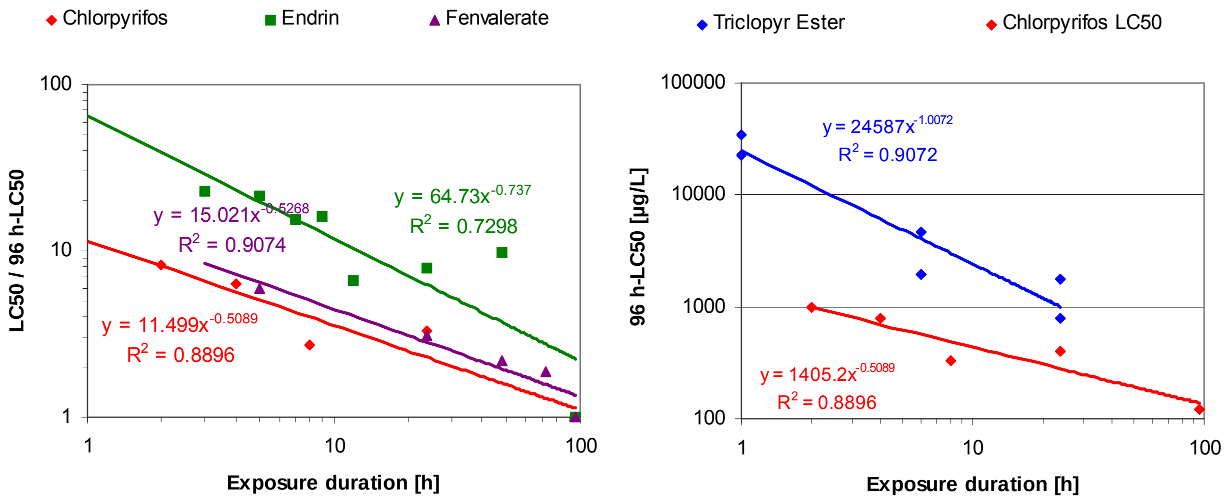

Figure 6-39: 96h-LC_{50} dependent on exposure duration for fish;
left: Data of Jarvinten et al. 1988 for *Pimephales promelas* – LC_{50s} related to the LC_{50} for 96 h exposure, right: 96h-LC_{50} of Kreutzweiser et al. 1994 for effects of Trichlorpyr ester on *Oncorhynchus mykiss* after 1. 6 and 24 h exposure (data for *P.promephales* and Chlorpyrifos of Jarvinten et al. 1988 are shown for comparison).

In addition, several datasets were available where the fish were exposed over 96 h, but survival was monitored also in between, usually after 24, 48 and 72 hours. These datasets can be divided into four groups:

Legierse (1998) tested 5 plant protection products by exposing guppies up to 14 day and daily monitoring their survival. The LC_{50} could be described by power functions with coefficients of determination from 0.87 up to 0.96 (Figure 6-40, left). However, the diagram indicates that the power function does not fit equally for short and longer exposure durations. Because the aim here was to extrapolate from the standard acute test to shorter exposure duration, in the right diagram only the data up to 96 h were used for the fit. While for azinphosmethyl, malathion and phosmet the fitted function was very similar, methidathion and phenthoate showed only a slight increase in LC_{50} if exposure was 48 or 24 h instead of 96 h.

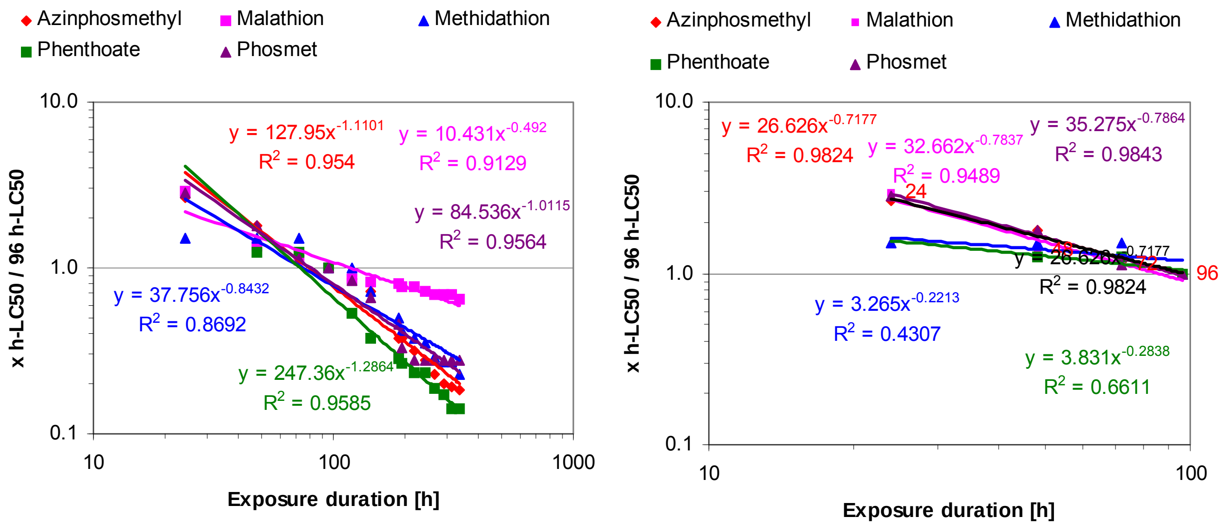

Figure 6-40: LC_{50} dependent on exposure duration for the guppy (Legierse (1998));
left: Full dataset, right: only data up to 96 h

Hunt (1981) analysed the effect of temperature on toxicity of hydrazine to the bluegill (*Lepomis macrochirus*) after 1, 6, 24 and 96 h. While the LC_{50} after 96 h was very similar (1 – 1.6 mg/L) the effect during the first 6 h was most pronounced at 21 °C but slowest at 15.5 °C. Because the values for 1 h exposure are all above the fitted line, fits without the 1 h-LC_{50} were calculated in the right diagram. The lowest slope was 0.42 found for 21 °C.

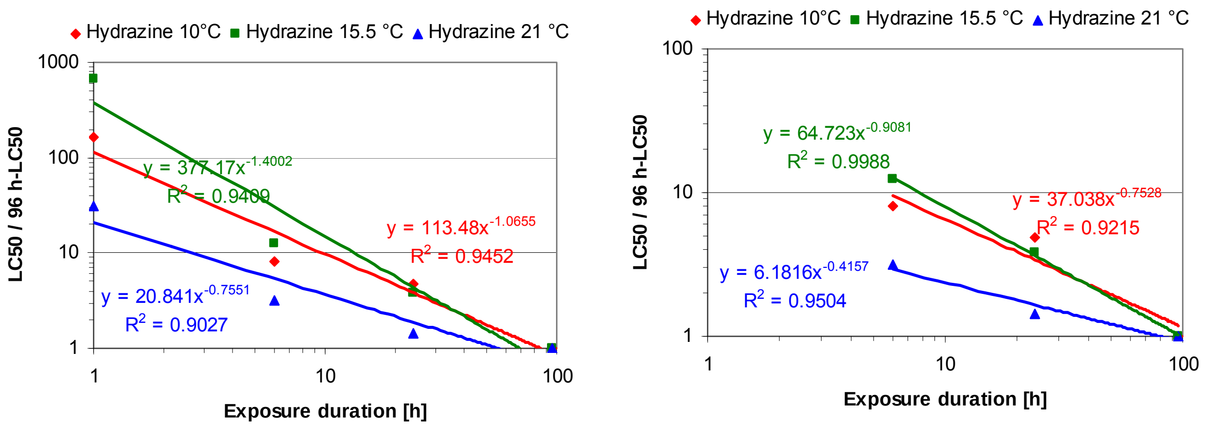

Figure 6-41: LC_{50} for the bluegill (*Lepomis macrochirus*) dependent on duration of exposure to hydrazine (Hunt (1981)) measured at three temperatures;
left: including the 1 h values, right: only values for 6, 24 and 96 h considered

Pickering & Henderson (1966) tested 12 chemicals with guppy, fathead or bluegill and in soft or hard medium. Not all combinations have been tested, but in total, 28 tests were conducted to provide LC_{50} after 24, 48 and 96 h. In most cases the LC_{50} did not decrease significantly after 24 h; the median ratio 24h-LC_{50}/96 h-LC_{50} was 1.2. The substances were named by codes only (e.g. American Cyanamid 12009, Tech.) but they all seemed to be organophosphates or carbamates. In some cases the same LC_{50} was reported after 24, 48 and 96 h. Thus, a generic worst case estimation for effects for exposure from 24 h to 96 h would be the 96 h estimation.

In the last group of tests discussed here, rainbow trouts were exposed to 5 chemicals:

Bailey et al. (1999) found no change in toxicity of didecyldimethylammonium chloride (DDAC) from 24 to 96 h while LC_{50} for 3-iodo-2-propynyl butyl carbamate (IPBC) could be fitted to a power function with a slope of 0.88.

A slightly less steep relation was obtained for deltamethrin data by Ural & Saglam (2005) for exposure duration from 1 to 96 h.

LC_{50s} for formalin and malachite green showed a similar relationship for exposure between 8 and 96 h in experiments of van Heerden et al. (1995). For malachite green additional observations are available for 4, 5, 6 and 7 hours which show that below 8 hours, the LC_{50} increases much steeper with shorter exposure time.

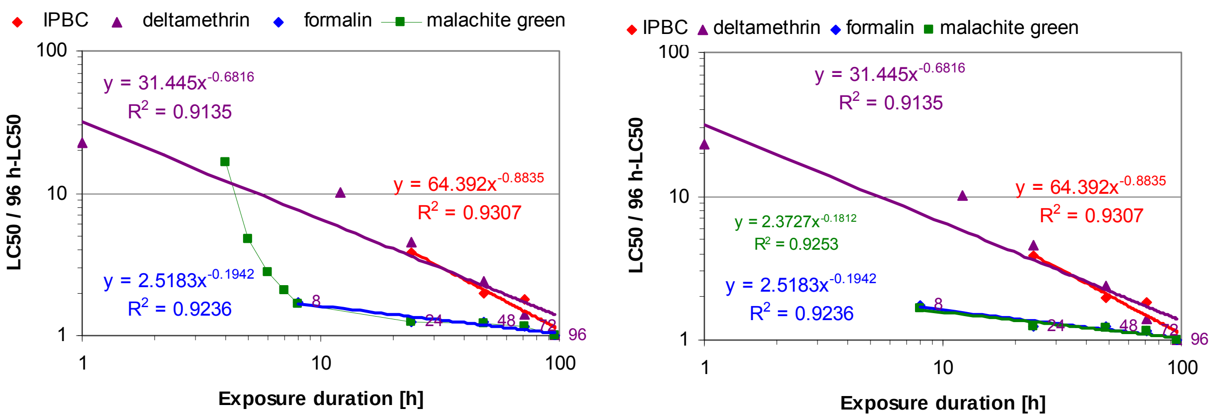

Figure 6-42: LC_{50} for the bluegill *(Lepomis macrochirus*) dependent on duration of exposure to hydrazine (Hunt (1981)) measured at three temperatures;
left: including the 1 h values, right: only values for 6, 24 and 96 h considered

In summary, the analysis of the available data on short-term exposure of fish show that for several substances a power function can describe the relationship between LC_{50} and exposure duration. However, there are some substances for which a steep decrease of LC within the first few hours was shown while later on mortality did not change with time. Thus, a generic worst case estimation for effects for exposure from 24 h to 96 h would be the 96 h estimation. For a more refined estimation, the data should be analysed based on the class of the active substance respectively their mode of action. The factors between 1 h LC_{50} and 96h LC_{50s} were determined to be:

- 11.5 for organophosphates (fathead minnow and chlorpyrifos) and
- 10.5 (malathion) – 128 in Guppy with daily observations only
- 15 – 31 for pyrethroids (lowest factor: fathead minnow and fenvalerate)
- 65 for endrin in fathead minnow.

6.3.2.8 Limitations of the approach

The approach developed here is only applicable for situations where the risk assessment is driven by acute toxicity to invertebrates or fish. For this situation, it provides a conservative estimation of an RAC* depending on the time over threshold, i.e. the PEC above the standard RAC. It is only applicable assuming single exposure events, because for time-variable exposure with multiple peaks other factors not included in this relationships play a role, e.g. carry-over toxicity (Ashauer et al. 2010, Preuss et al. 2008 & 2009). For multiple exposure events mechanistic modelling approaches, taking the toxicokinetics and toxicodynamics into account, have to be used to get a conservative prediction of the effects on individual as well as on population level (Ashauer et al. 2010, Preuss et al. 2008, 2009).

For chronic endpoints, e.g. NOEC for inhibition of reproduction, growth and development or population growth rate (algae, *Lemna*), we suggest to decide case by case if the duration of exposure is sufficient to induce a relevant effect. If so, the recommendations of the eLink workshop (Brock et al. 2009) should be followed, i.e. to calculate PEC_{TWA} over a relevant time span to be compared with the exposure in the chronic toxicity test.

If the risk assessment is driven by a NOEC or NOEAEC from mesocosm study following a static exposure regime, in the first step the RAC should be compared to the $PEC_{initial}$ respectively the maximum PEC_{TWA} over 1 h. If the predicted exposure is significantly shorter than measured in the mesocosm study, the TWA approach could be used as for chronic tests. See also the recommendations of the eLink workshop (Brock et al. 2009).

6.3.2.9 Conclusions

The following suggestions for the consideration of exposure significantly shorter than in the standard tests are made:

- If the effect assessment is based on the **acute Daphnia test** over 48 h:
    - If the PEC is above the RAC for less than 24 h, the substance specific EC_{50} for 24 h should be used. Because uncertainty regarding latency of effects is not changed compared to the standard test, the trigger value of 100 should be applied.
    - If only the 48h EC_{50} is available, a more realistic but still conservative estimation of a RAC^* for short exposure (< 24 h) could be done by multiplying the standard EC_{50} by 1.6 (the median of the data analysed here) and keeping the standard trigger value. No results for effects on *Daphnia* after less than 24 h were available yet.
- If the effect assessment is based on **acute tests with invertebrates** over 96 h:
    - The RAC* for pulse exposure can be estimated by the following formula based on data for carbaryl shown to be protective for other substances:

$$RAC_t = RAC_{96h} * 5.05\ t^{-0348}$$

- If the effect assessment is based on **acute tests with fish** over 96 h:
    - For some substances no differences for 24 and 96 h – LC_{50} were found.
    - The factors between 1 h LC_{50} and 96 h LC_{50s} were determined to be:
        - 11.5 for organophosphates (fathead minnow and chlorpyrifos) and

    - 10.5 (malathion) – 128 in Guppy with daily observations only
    - 15 – 31 for pyrethroids (lowest factor: fathead minnow and fenvalerate)
    - 65 for endrin in fathead minnow.

- For **chronic endpoints**, e.g. NOEC for inhibition of reproduction, growth and development or population growth rate (algae, *Lemna*), the recommendations of the eLink workshop (Brock et al. 2009) should be followed, i.e. to decide if and for which time window the time weighted average (TWA) concentration should be used.
- If the risk assessment is driven by a NOEC or NOEAEC from a **micro- or mesocosm study:**
    - First the RAC should be compared to the $PEC_{initial}$ (respectively the maximum PEC_{TWA} over 1 h from the dynamic model).
    - If this indicates a risk and the predicted exposure is significantly shorter than measured in the study, the TWA approach could be used as for chronic tests, see the recommendations of the eLink workshop (Brock et al. 2009).

6.4 Consideration of multiple applications

The issue of multiple applications of a product is not relevant for the generic analysis aiming to identify management segments because this would not change the ranking of the water body segments or the spatial distribution according to their pesticide entries.

However, for authorization of a specific plant protection product it might be necessary to consider multiple applications according to the GAP of the product.

Therefore it was calculated how large the effects of the single applications could be to result in a given total effect <u>under the assumption of independent effects of the single applications</u> (i.e. no increased sensitivity of pre-exposed organisms but also no recovery of the populations between the applications). The following table provides the resulting **adapted tolerable effect levels** to be used in the hotspot criterion for the single application for different numbers of applications and different levels of total effects.

For, example, if in total 20 % reduction of a population are considered to be tolerable, each of three application should not exceed 7.2 % effect. Thus, 100 % - 7.2 % = 92.8 % would survive the first application, and from these again 92.8 % would survive the second application (thus, in total, 92.8 %² = 86.1 %. Again, 92.8 % from these would survive the third application, resulting in 92.8 $\%^3$ = 79.9 %.

Table 6-10 Tolerable effect levels for the single application for different levels of total effects and different numbers of applications per year under the assumption of independent effects of the single applications. Tolerable effect [%] for a single of n applications (tol_eff_n calculated as $tol\_eff_n = 100 – (100 –tol\text{-}eff_1)^{1/n}$

n_appl	% tolerable effect								
1	10.0	20.0	30.0	40.0	50.0	60.0	70.0	80.0	90.0
2	5.1	10.6	16.3	22.5	29.3	36.8	45.2	55.3	68.4
3	3.5	7.2	11.2	15.7	20.6	26.3	33.1	41.5	53.6
4	2.6	5.4	8.5	12.0	15.9	20.5	26.0	33.1	43.8
5	2.1	4.4	6.9	9.7	12.9	16.7	21.4	27.5	36.9
6	1.7	3.7	5.8	8.2	10.9	14.2	18.2	23.5	31.9
7	1.5	3.1	5.0	7.0	9.4	12.3	15.8	20.5	28.0
8	1.3	2.8	4.4	6.2	8.3	10.8	14.0	18.2	25.0

As for the total tolerable effect levels, the tolerable effect levels for the single application events can be refined by a higher tier risk assessment, e.g. considering dissipation of the active substance and the recovery potential of the relevant taxa.

On the exposure side, this approach would require the following assumptions:

For the **static exposure model** (for lentic systems, based on the PEC_{ini}) it should be considered that for multiple applications it is unlikely that each application results in entry and thus PEC_{ini}, equal to or above the 90^{th} centile of the local distribution. This is also considered in the current approach where reduced centiles are used to calculate the PEC_{ini} from the sum of the entries of the single applications (FOCUS 2001, Table 6-11):

Table 6-11: Percentile of individual spray drift events for n applications which are equivalent to cumulative 90th percentile spray drift for the season (Table 5.4.2-1 in FOCUS 2001)

Number of applications	Drift percentile of a single event	Cumulative drift percentile for the season
1	90	90
2	82	90
3	77	90
4	74	90
5	72	90
6	70	90
7	69	90
8	67	90
>8	67 (assumed)	90+

In contrast to the FOCUS approach where the single event PEC_{ini} is multiplied with the number of applications for the calculation of the TER, here the effect of the multiple applications is considered on the effect side by the reduced tolerable effect thresholds listed in Table 6-10. Thus, the PEC_{ini} of a single event should be calculated based on Table 6-11.

The **dynamic exposure model** used for PEC calculations in lotic waters per se provides time series of the PEC for each water body segment. Thus, it seems to be straight forward to model the whole application period and to extract the maximum $PEC_{TWA(1h)}$ for the further assessment using the adapted tolerable effect level. In a first step, the total ToT should be used for the calculation of the RAC_{dyn}.

As said at the beginning this approach to consider multiple applications by adapting the tolerable effects is based on the assumption that the effects of the single application events are independent.

This might be not conservative if pre-exposed individuals are more sensitive than non exposed ones which is the case when individuals have not recovered from the exposure. If this is the case, the single events would result in higher effects than calculated in Table 6-10.

On the other side, the approach also ignores potential recovery between the application events which can be relevant for species with continuous reproduction, e.g. algae and many zooplankton species.

As and example, simulated effects of 1 to 5 applications resulting in the same total effect (90 %) according to Table 6-10 are shown in Figure 6-43. Of course recovery within the application events can significantly reduce the maximum reduction of abundance and the time necessary to recover close to control levels again.

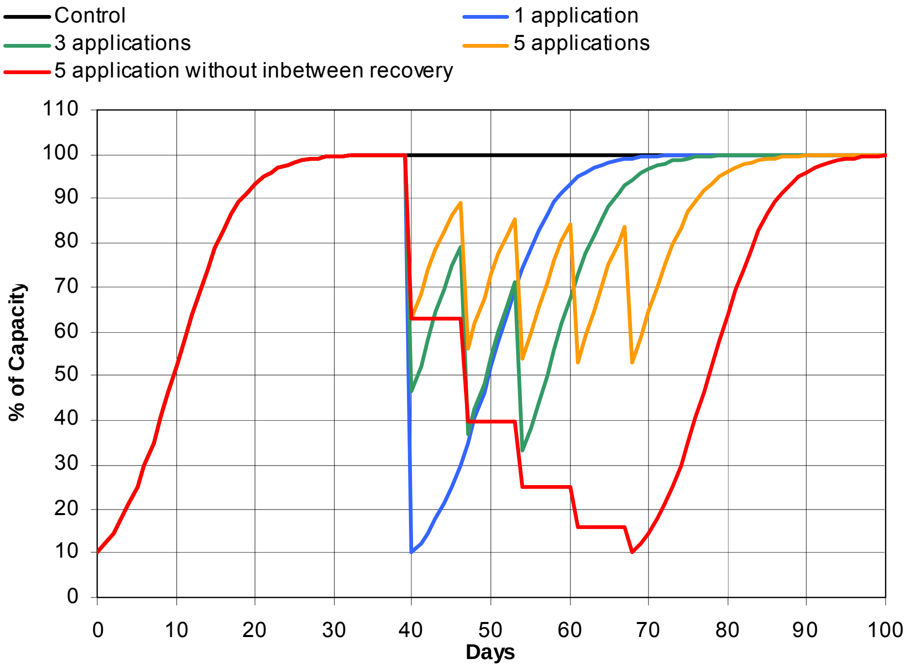

Figure 6-43: Effects of one to five pulse exposure events on a simulated rotifer population on a rotifer population with theoretically the same total effect (90 %). The logistic model used here was based on Barnthouse (2004) and on Hommen (2008, unpublished report).

For species with discrete reproduction events as given for the realistic worst case species in chapter 6.3, the effect of multiple applications depends on the timing of the application related to the time of reproductions. Only if the reproduction falls within the application period intrinsic recovery can take place between applications. Otherwise, there is only the possibility for external recovery (which was neglected in the derivation of the tolerable effect levels). The importance of recolonisation might be species dependent – nearly unimportant for less mobile species but important for mobile species where reproduction is depending on a territory which can be quickly occupied. Thus, also here, the effects of multiple applications can assumed to be additive.

However, the modified tolerable effect classes of Table 6-10 do not consider the prolonged duration for recovery compared to one application with the same effect. Without recovery within the application period, the recovery time is prolonged by the duration of the application period.

A detailed analysis of the effects of multiple applications on the dynamics of the realistic worst case species selected in chapter 6.3 has not been conducted yet.

For a more refined assessment, the dissipation of the substance between applications could be considered to estimate the effects of the different applications. This and more detailed assessments, e.g. by considering recovery in between application and possible carry over effects (the same dose might have higher effects with increasing number of applications, e.g. Ashauer et al. (2010), Preuss et al. (2008 & 2009) are not within the scope of this project.

6.5 Support of the technical implementation of the hotspot definition

It has been discussed with WP 2 how the hotspot criteria can be implemented in the final tool for the geodata based probabilistic risk assessment.

For the implementation of the hotspot criteria in GIS a moving window approach was suggested by the UBA (2007): A window of e.g. 1000 m length is moved segment per segment over a water body in downstream direction. At each step, it is checked if the hotspot criterion is fulfilled, e.g. for the generic criterion, if the PEC in 10 % or more segments is above the RAC. Thus, in this example, a hotspot is at least 1000 m long but might be longer.

From a technical point of view there are a few problems regarding the quality of the geodata and the moving window approach:

- ATKIS does not provide a connected network of the water bodies but a large number of artificial short sections
- ATKIS does not provide flow directions
- ATKIS might always provide correct distances between crop and water body (unused areas are often classified as agricultural areas).

Therefore, the potential use of the hotspot criteria can only be analysed by the use of high-resolution data (HR data) providing a directed and connected web of water bodies, realistic distances between crop and water body and information on drift reducing vegetation.

WP 2 has developed a web tool for the geodata based probabilistic exposure estimation including the generic hotspot criterion of 10 % risk segments in 1000 m water body length. Analyses of WP 3 have shown that this criterion is reliable for a generic risk assessment. However, results of WP 2 have shown that this criterion results in an only slightly lower number of management segments compared to the number of risk segments if the assessment is based on the static model alone (without applying the hotspot criterion).

The only refinement of the generic hotspot criteria used in combination with the static exposure model would be the replacement of the assumption that there is directly 100 % effect if the PEC is above the RAC. In chapter 6.3 a slope of 4 in the logistic dose response relationship was suggested as a realistic worst case slope to calculate effect per segment from the PEC (assuming that the RAC can be considered to be the EC_{10}).

For the dynamic model it has been shown that it is necessary to consider the often very short exposure in flowing waters (chapter 6.3).

Within the project WP 3 has suggested input and output forms with respect to the hotspot analysis:

The input form is divided into two sections. In the first section, the (aquatic) ecotoxicological profile of the product is summarized and RAC are calculated for each test type using standard or (reduced) higher tier triggers.

In the second part, the hotspot analysis is parameterized. For each of the trait based groups finally defined (section 6.2[45]), the most relevant toxicity endpoint should be selected. In addition the assessment factor for deriving an RAC must be entered, depending on the uncertainty on extrapolating from the study endpoint to the trait group. For example if some data for the acute toxicity of fish are available, the uncertainty on using these data for the trait group fish can be considered to be small. On the other side extrapolation the toxicity of fungicide from a *Daphnia* test to a trait group representing non-arthropods like molluscs might be large and the standard assessment factor could be appropriate.

The relevant length of the water body section and the tolerable effect magnitude will be fixed once the hotspot criteria have been finalized.

Then for each of the (relevant [46]) trait based groups the hotspot criteria should be applied and again number of remaining hotspots as for the output of results, WP 3 has suggested to provide a table comprising information on the number, length and proportion of potential RMS for each of the RAC derived in the ecotoxicological profile in order to give an impression of the frequency of predicted RAC the final criterion for the risk assessment and as useful information especially for generic analysis, the statistics on the RMS should be provided.

Other types of outputs, e.g. cumulative distributions of the PECs on a landscape or regional level or maps of PECs or TERs might also be useful.

Note that it was finally concluded that the dynamic exposure model should be used for a more realistic risk assessment considering the properties of the water bodies in the landscape. Therefore it would be necessary also to offer the possibility to consider the exposure duration and the tool should allow estimating effects depending on the exposure duration.

As an effect of the parallel work in the different work packages, not all suggestions of WP3 could be implemented in the web tool developed in AP2. In the current state, the web tool uses a fixed hotspot criterion of 10 % tolerable effect in 1000 m moving window (water body length). It is not possible yet to modify the tolerable effect threshold, nor to consider multiple applications. Effects in a segment were assumed to be 100 % if the PEC exceeds the RAC; a generic or a substance specific dose-response function was not implemented yet.

[45] Which are now: phytoplankton, zooplankton , ‘worm,’ ‘dragonfly’, ‘clam’, fish, macrophytes, see chapter 0

46 For a generic assessment, the most stringent criterion of 10 % tolerable effect should be applied. For a specific product, the criterion of the most relevant group could be used. For example, it would usually not be necessary to conduct a hotspot analysis for algae and macrophytes for an insecticide. Thus, if it can be demonstrated that insects are the most sensitive group (with respect to intrinsic toxicity) the criterion for the dragonfly could be used, if algae are the most sensitive group (and macrophytes are clearly less sensitive) the more relaxed criterion for algae could be applied.

Table 6-12: First suggestion of WP 3 for input forms in the tool developed by WP 2 (status 2009)

Ecotoxicity profile of the product							
No.	Test	Endpoint	days	Conc. [µg/l]	Trigger	RAC	Ref. / comment
	Standard acute						
1	Daphnia acute	EC50	4	20	100	0,2	--
2	Other Invertebrate acute	EC 50	4	10	100	0,1	--
3	Fish acute (most sens. of 2 species)	LC50	4	100	100	1	--
	Standard long-term (not all tests always required)						
4	Algae (most sens. of 1 or 2 species)	EC 50	4	500	10	50	--
5	Lemna	EC 50	7		10	0	--
6	Daphnia reproduction	NOEC	21		10		
7	Chironomus emergence (water)	NOEC	28				
8	Chironomus emergence (sed.)		28				
	Higher Tier (case by case)						
9	Invertebrate SSD	Geomean	4	70	100	0,7	--
10	Fish refined exposure	LC50	4	600	100	6	--
11	Mesocosm	NOEC	56	50	2	25	
12	Mesocosm	NOEAEC	56	200	5	40	
13	optinal other... higher tier results						
14							
15							

Hot spot analysis - identification of real RMS						
Group	Test used	AF	RAC	Length (m)	Tol. Eff. %	
Generic	most sensitive	100	Tox / AF	1000	10%	
Algae	1			x	y	
Macrophytes	1			x	y	
Zooplankton	3			x	y	
Benth. Crust	3			x	y	
Insecta 1	4			x	y	
Insecta 2	4			x	y	
Mollusca	4			x	y	
Fish	5			x	y	
maybe other group	5			x	y	
maybe other group	5			x	y	

Table 6-13: First suggestion of WP 3 for output forms in the tool developed by WP 2 (status 2009)

Results: actual RMS and hot spots				
Group	% RMS	n RMS	length RMS	**n hot spots**
Generic				
Algae				
Macrophytes				
Zooplankton				
Benth. Crust				
Insecta 1				
Insecta 2				
Mollusca				
Fish				
maybe other group				
maybe other group				

6.6 Summary of the assumptions for the proposed hotspot criteria

Corresponding to Table 4-23 summarizing the assumption for the exposure assessment, the following table gives an overview on the effect and risk assessment.

Table 6-14: Comprehensive overview on parameters and variables used for the effect assessment and hotspot identification

Variable, parameter	**Probabilistic/ deterministic**[a]	**Protectivity, percentile**[b]	**Geo-referenced**[c]	**Methodology, value(s)**
Relevant exposure characteristics	d	yes[b]	yes	Static water body: $PEC_{ini;}$ Dynamic water body: max $PEC_{TWA(1h)}$, ToT (Time over threshold); protective because dissipation not considered
RAC	d		no	Derived as in the current approach from ecotox tests / SSDs / mesocosm studies using safety factors (triggers); assumed to be protective
Effect per segment = % reduction of abundance	d	yes[b]	no	Logistic dose response function effect % = $100/(1+(PEC/EC50)^{-slope}$ EC_{10} = RAC slope = 4 for generic assessment slope from ecotox test for product risk assessment, EC50 = f(slope, EC_{10}) not implemented in the web tool (AP2) yet (here 100 % for PEC>RAC is assumed)
RAC_{dyn}	d	yes[b]	yes	Estimation of lethal effects of short term exposure RAC_{dyn} = empirical functions of RAC and ToT protective due to use of data for worst case substances; not included in the web tool yet (because based on PECini)
Spatial scale for hotspot criterion considered to be conservative	d	yes	no	1000 m (length of moving window) protective because larger scale would result in less hotspots
Tolerable effect level for hotspot criterion	d	yes	no	Generic 10% (as used in the web tool) Should be depend on taxon, e.g. 10 % for macroinvertebrates and fish, 90 % for phyto and zooplankton, and number of applictions; Refinement by higher tier tools possible
Multiple applications	d		no	Tolerable effects levels for single application events based on number of applications and total yearly tolerable effect Protectivity not quantified, single events considered to be independent Not implemented yet.
Hotspot criterion	d		no	Total effect within 1000 m water body above tolerable effect level
Consideration of sublethal effects	d	yes[b]	no	Considered as lethal effects (reduction of abundance)
Effects of entries of other products	-	no	-	As in the current approach, not explicitly considered
Recolonization (over larger distances, e.g. from river not directly connected and/or from tributaries)	-	yes[b]	-	Not regarded (not applicable for generic trait based approach); Protective because recolonisation would increase recovery

a) b) c) refer to Table 4-23

6.7 References

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7 Identification of potential management segments for water bodies close to permanent crops in Germany using the static exposure model

Burkhard Golla, Jens Krumpe

7.1 Implementation of the potential MS identification for GeoRisk static

The GeoRisk hotspot analysis bases on the GeoRisk Exposure Database and forms the end of the computing workflow of the GeoRisk static approach. The intention was not to consider the exposure situation at one single water body segment but to expand the spatial level of analysis on the level of populations by analysing connected water body segments with "moving window". It implements the conceptual proposal of UBA for stagnant water bodies (UBA 2007) because analyses of WP3 have shown that this hotspot criterion (10 % effects in 1000 m water body, see chapeter 6.1.1) is reliable for a generic risk assessment (see chapter 6). Due to parallel work in the different work packages, some suggestions of WP 3 (chapter 6) could not be considered in the implementation of the web tool yet (see chapter 6.5 and 6.6). For example, the tolerable effect level was fixed to 10 %. A variation of the effect level according to the number of applications was not implemented yet and the effects per segment with PEC > RAC were assumed to be 100 % without applying a generic realistic worst case dose-response function.

Figure 7-1: Hotspot analysis bases on the GeoRisk exposure database

The technical implementation remained close to the conceptual proposal. The criterion "*a hotspot is given …if at least in one segment a PEC higher then ten times the RAC*" (UBA 2007) was not implemented because a buffer zone (no spraving zone) of 10 m along the water bodies was assumed and the PEC for 10 m can not be 10 times higher than the PEC for 20 m which is used as the RAC in this generic assessment. The concept of a "moving window[47]" is realised within a network data model (NDM) which lets users model and analyze networks such a stream networks.

Before the Germany wide analysis can be conducted the BDLM water bodies need to be transferred to network topology. A comprehensive discussion about the BDLM short comings is given in Koschitzki (2004) and chapters 6.3, 8.2 and the technical guidance document. As e.g. the BDLM do not provide hydrological correct flow direction data a network analysis is conducted up

[47] „moving window" according to the UBA proposal (2007) does not explicitly refer to a raster analysis method but is used to describe the approach of analyzing connected water body segments according to given criteria

and down stream from any risk segment (RS) assuming that ditches have no predominant flow direction (Figure 7-3). The network analyses implements Dijkstra (1959) graph theory[48].

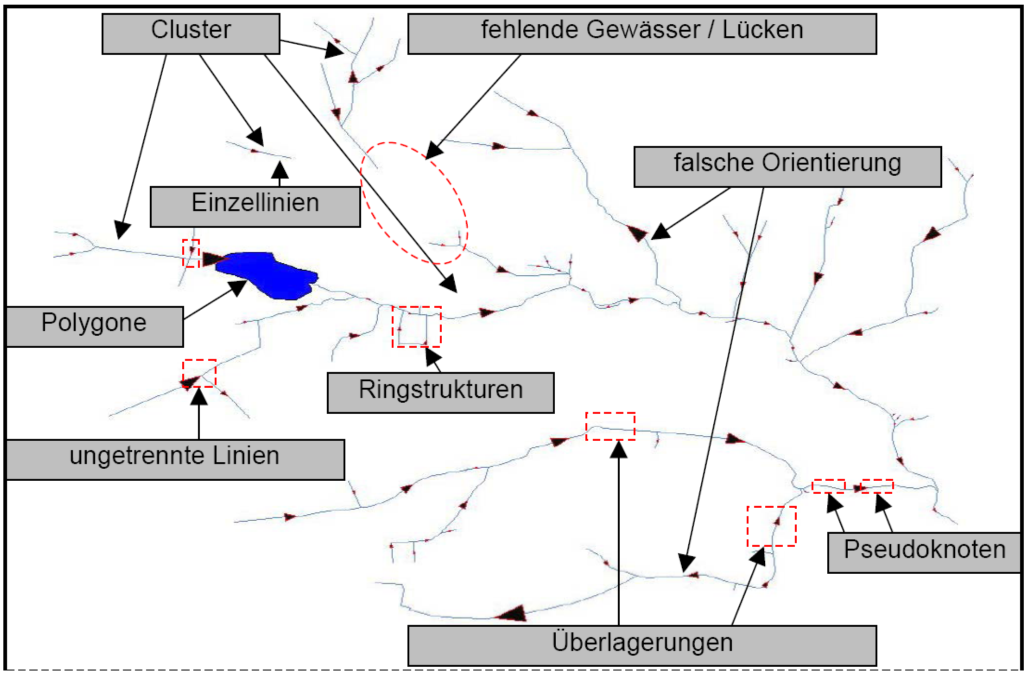

Figure 7-2: Common errors and obstacles in transferring BDLM watercourse data into hydrological correct network topology (after Koschitzki 2004)

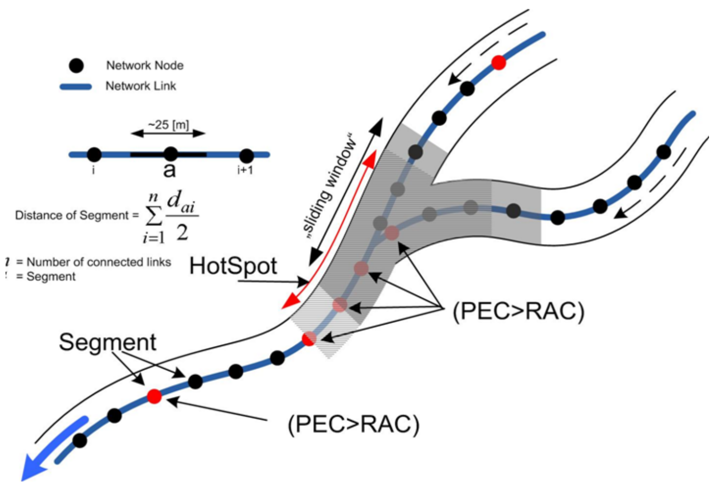

Figure 7-3: Concept of the GeoRisk hotspot identification for stagnant water bodies

[48] shortest path algorithm

7.2 Results of the hotspot analysis for a GeoRisk static assumption

The following tables and figures show the results of the RS computation and MS analysis for the static GeoRisk model according to the UBA proposal (UBA 2007). The width-depth ratio for the exposure calculation is 3.3 : 1. Field data (see chapter 4.4) indicate, that the width to depth ratios of stagnant ditches are usually larger. However, it was decided not to change the up to now established ratio for GeoRisk static calculations unless a more expanded and regionalized database would give solid arguments to do so. Thus, the following results were obtained for the current (e.g. FOCUS, Exposit) standard ditch scenario with a ratio of 3.3 : 1.

Meanwhile the results of chapter 5.4.4 on the sensitivity of a higher width (depth ratio (6.6 : 1) can be used to estimate the effects also on a national scale. The maps show the summarized length of management segments at tile level of the topographic map (TK25) with a general coverage of ca. 120 to 140 km².

7.2.1 Results of the hotspot calculations for hops

The results for hops clearly point to the location of the growing regions which are located in Bavaria and in a much lower extend Baden-Württemberg and Thuringia. Management segments are located in 57 TK25 tiles. Looking at the length of management segments on tile level: 50 % of the tiles contain less then 600 m of management segments per tile, 10 % contain more then 9.1 km with a maximum of 23.8 km per tile.

7.2.2 Results of the hotspot calculations for vine

The results for vine show a similar picture. In the federal states with the main growing regions, Baden-Württemberg and Rhineland-Palatinate, most management segments are located. Management segments are found in 194 TK25 tiles. 50 % of the tiles contain less then 660 m of management segements per tile, 10 % contain more then 9.4 km per tile with a maximum of 57 km per tile.

7.2.3 Results of the hotspot calculations for fruit

The results for orchards differ considerably. Many scattered tiles with management segments reflect the widespread existence of orchards. Management segments are found in 752 TK25 tiles. 50 % of the tiles contain less than 350 m of management segments per tile, 10 % contain more then 3.1 km per tile. Three tiles in growing region *Altes Land* contain more then 70 km per tile.

Table 7-1: Results of the hotspot calculations for hops

Region	Water bodies in theoretical drift zone [km]	MS [km]	RS [km]
Hops Germany	1 019	141	146
BW			19.39
BY			112.77
RP			0.18
SN			3.00
ST			0.01
TH			6.63

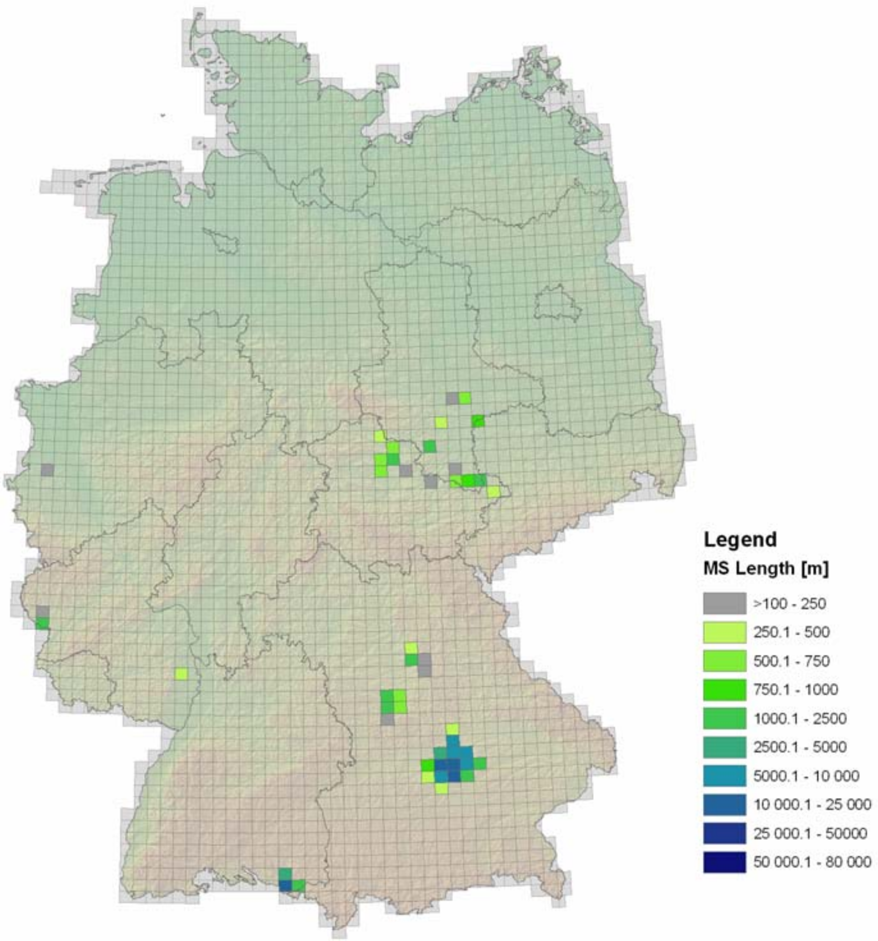

Figure 7-4: Geographic density of management segments for hops aggregated on TK25 tile level

Table 7-2: Results of the hotspot calculations for vine

Region	Water bodies in theoretical drift zone [km]	MS [km]	RS [km]
Vine Germany	3 432	657	663
BW			133.2
BY			32.1
HE			52.8
NW			0.6
RP			442.5
SL			0.3
SN			0.4
ST			0.8
TH			0.1

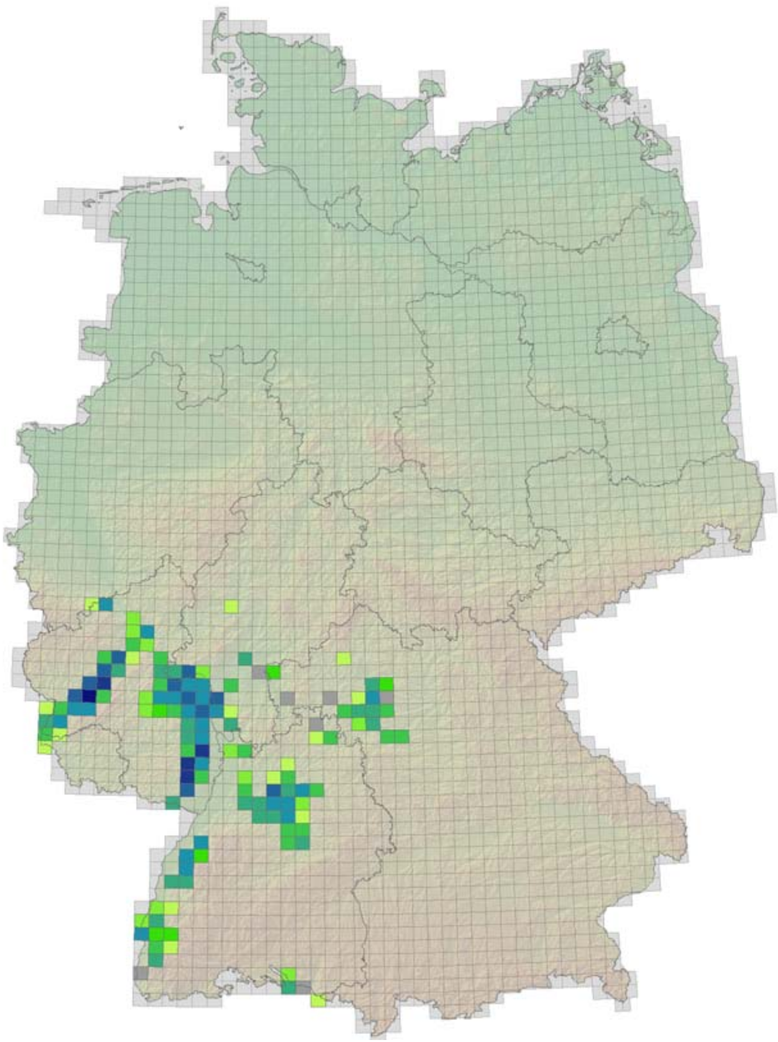

Figure 7-5: Geographic density of management segments for vine aggregated on TK25 tile level

Table 7-3: Results of the hotspot calculations for fruit

Region	Water bodies in theo-retical drift zone [km]	MS [km]	RS [km]
Fruit Germany	7 334	2483	2.518
Only *“Altes Land”*	1 640		1.259
BB			13.1
BW			432.8
BY			87.5
HE			21.4
MV			29.4
NI			1068.7
NW			43.2
RP			92.9
SH			572.1
SL			0.6
SN			20.7
ST			116.7
TH			18.8

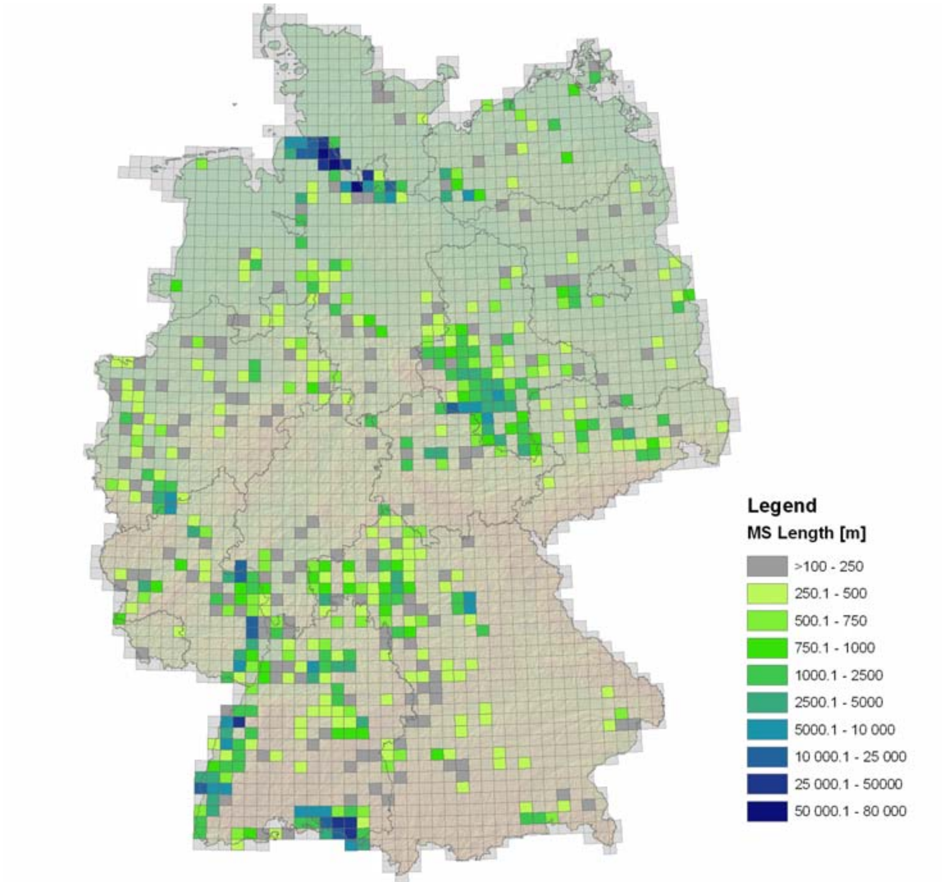

Figure 7-6: Geographic density of management segments for orchards aggregated on TK25 tile level

7.3 Conclusions

The following conclusions can be drawn from these results:

- The underlying criteria of the implemented hotspot approach show little influence on the total length of management segments. The approach reduces the amount of risk segments of 0.9 % for vine, 1.4 % for orchards and 2.7 % for hops. The majority of risk segments remain part of management segments as most application sites that are located directly adjacent to surface waters share a neighbourhood of at least more then 100m. The sensitivity of the parameters tolerable effect and the length of the *"moving window"* on the total length of MS different scenarios were analyzed by WP2 (chapter 4.2.2) and WP3 (capter 6.2.1). The results at which "window" length and tolerable effect the amount of management segment significantly decreases are presented in chapter 6.2.6.
- The geographic density of management segments for vine and hops clearly show the location of the main growing regions. It counts for all permanent crop types, that there are no individual geographic locations inside the growing regions where management segments cluster. Whenever an application site is in a near distance to water bodies it is detected as a management segment according to the GeoRisk static approach. This gets obvious looking at the situation for orchards. A lot of tiles with a total length of management segments above 500 m are scattered throughout. For a first verification based on image analysis as proposed in chapter 5.6 this is not a handicap but could become an obstacle for verification in the field due to the distance between the locations.

7.4 References

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8 Identification of potential management segments for selected water bodies close to hop fields using the dynamic exposure model

Matthias Trapp, Kai Thomas, Djamal Guerniche

8.1 Introduction

Chapter 8 describes the examplary application of the dynamic exposure model to selected streams in the hops region Hallertau and the orchard region at Lake Constance. Information on the general concept of the calculation of drift entries into water bodies is given in sections 4.1 to 4.4. The theoretical concept of the dynamic exposure model is outlined in section 4.5.

8.2 Generating a consistent orographical stream network

The integration of the above mentioned dynamic model into a GIS environment requires a line-based geodataset representing a consistent water course stream network including its orographical conditions. The raw line-based ATKIS water course dataset does not fulfil these requirements. So it was necessary to develop a number of GIS-based algorithms in order to correct the data and meet the needs of the project.

Based on the use and modeling of digital terrain models these algorithms were implemented in ArcGIS and are now ready for use. This work was done as a diploma thesis related to the GeoRisk project (Guerniche et al. 2009).

8.2.1 Methods

The process of completion and correction is a tiered approach with several algorithms each based on the previous one:

- Alignment of the line-based ATKIS water course dataset using a digital terrain model (DTM)
- Bridging gaps by using the flow direction
- Realignment of the line-based ATKIS water course dataset using information of the receiving water courses

8.2.1.1 Alignment of the line-based ATKIS water course dataset using a digital terrain model (DTM)

The first algorithm assigns altitude information (source: DTM) to the vertices of every single corresponding line object of the water course dataset. The second and third algorithms separate sloped from non-sloped stream segments. A flow direction is assigned according to the altitude of the line vertices where possible. Non-sloped segments are marked for further processing..

8.2.1.2 Bridging gaps by using the flow direction

The backbone of the approach solving the problem of gaps in the water course network is the so-called flow direction, a DTM-derivative. With this information it was possible to obtain the most likely pathway of the missing segment and then link the loose ends of the stream step by step (Figure 8-2).

32	64	128
16		1
8	4	2

Figure 8-1: Codification of the flow direction raster (e.g. value "4" represents "south", "64" represents"north")

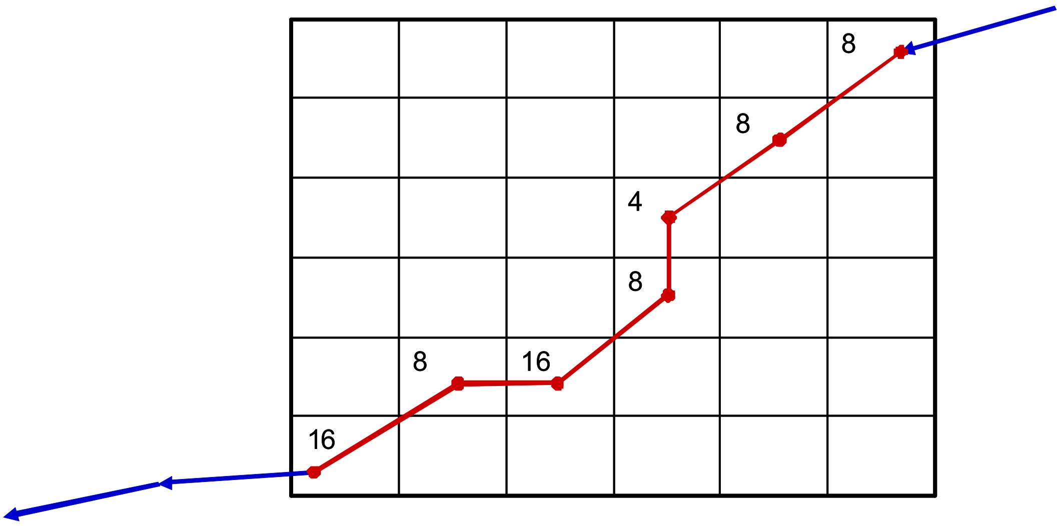

Figure 8-2: Schematic overview of the functional principles of the gap-filling algorithm. Grid: flow direction raster data, blue: ATKIS water courses, red: newly added segments (coding see Figure 8-1)

8.2.1.3 Realignment of the line-based ATKIS water course dataset using information of the receiving water courses

Since the two previously described algorithms did not work properly in landscapes without significant differences in altitude, an additional step was developed using neighbourhood information instead of flow direction values.

The main assumption here is that all water courses lead downwards to the next receiving water body. In an iterative process beginning at the junction with the last (usually lowest) receiving water body the newly developed algorithm upwardly checks every line segment concerning its flow direction. In case of incorrect alignment the segment is turned around.

8.2.2 Results

The result of the refinement of the raw ATKIS dataset is a consistent orographical stream network.

Sensitivity analysis showed a correct processing in 95 % of occuring gaps and incorrectly aligned line segments.

At this point some remarks on the aims of the processing and the accuracy of the now "consistent" water course network are necessary. The main concern of these processing steps was the generation of a dataset facilitating the unhampered functioning of the above mentioned dynamic model. Secondly, the work intended to create a reproduction of the real water course network as exact as possible. Due to many influences and restrictions in the underlying data this turned out to be difficult. The approach predicts the ideal pathway that water may take following the channel network (valley bottom centerlines) of the DTM. As the DTM used here has a ground resolution of 25 m the estimated water course can differ significantly from the course of the natural (real) water body. Additionally human activities modify the network of water bodies especially in densely populated or intensively used areas. Here, too, the real water bodies oftenly stray off the course of the valley bottom centrelines.

The following two figures show examples of the raw data and the same dataset after the refinement.

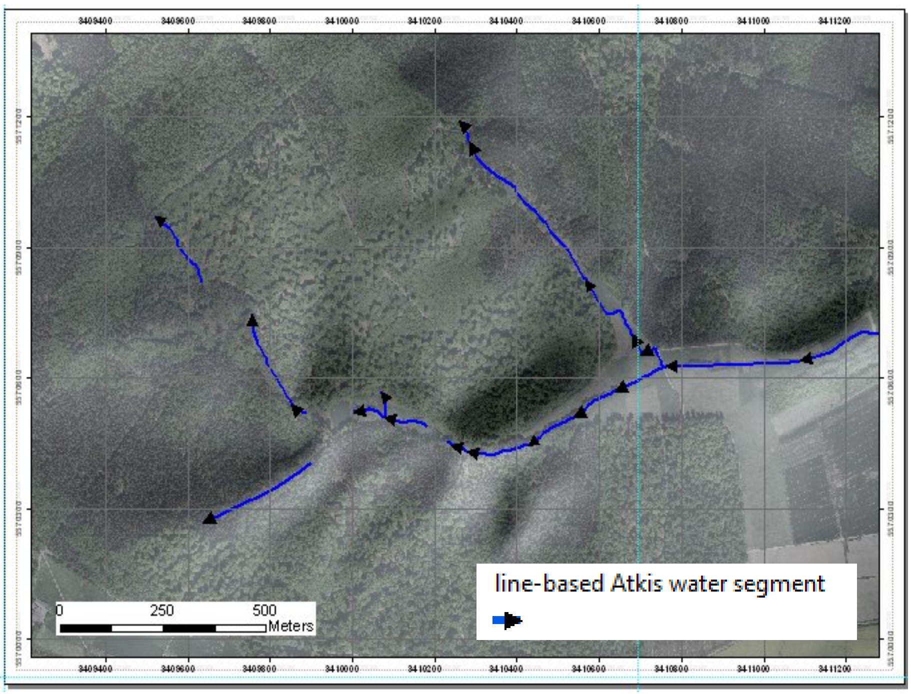

Figure 8-3: Map of the raw line-based ATKIS water body dataset

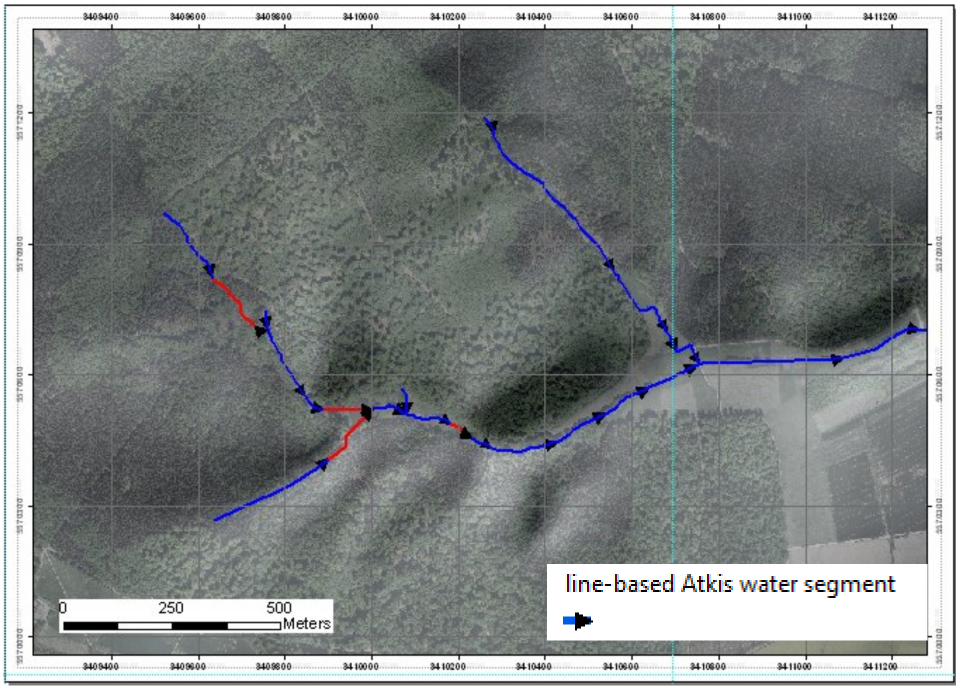

Figure 8-4: Map of the refined line-based ATKIS water body dataset without gaps and with the realignment of the flow direction of the segments

It can be summarized that in principle the approach introduced here can facilitate the refinement of a stream network for the whole of Germany. Hence it provides the requirements for a nationwide dynamic modelling of pesticide entries into surface water bodies. Nevertheless, the restrictions of the underlying data (DTM) and their influence on simulations have to be taken into account. In addition the application of the algorithm which makes use of the receiving water bodies as direction "guides" for the realignment of segments in non-sloped areas is not yet tested for greater regions.

This refinement was realised in the hop region Hallertau to facilitate the dynamic modelling of selected streams.

8.3 Geo-referenced dynamic modelling of longitudinal dispersion of the initial deposition in flowing systems

This new approach was developed due to the fact that the majority of surface water bodies in Germany are flowing waters whereas the current method for calculating the PEC is based on the concept of stagnant water bodies.

There are yet some methods of modelling the effects of flowing waters in Germany (Great-ER, s. http://www.great-er.org), but all these approaches have been developed for large streams like the river Rhine or the river Main, and not for small streams.

Detailed surveying in the hop region Hallertau as well as in the orchard region Lake Constance showed that the majority of the relevant water courses are small brooks.

Therefore, it was necessary to refine the existing methods and adapt them to the modelling of small brooks.

The mathematical concepts and the hydrological backgrounds are described in detail in the chapter 4.5.

In the following chapter the implementation in ArcGIS is described.

8.3.1 Mathematical concepts

The simulation of dilution of the initial PEC (PEC_{ini}) due to dispersion processes along the flow path is realized by the "PECdispers-Tool" (Wagner et al. 2007, Trapp et al. 2008b) for conservative substances (substances that do not degrade).

The $PEC_{dispers}$-Tool

The "$PEC_{dispers}$-Tool" simulates the spatiotemporal evolution of initial deposition in water body segments. Therefore the time period from the entry time to a certain point in time (relevant time period) is discretized and divided into equal time units (sampling interval: Δt). For each discrete time unit and each water body segment that the entry water package traverses while it flows, the PPP-concentration ($PEC_{dispers}$) is calculated by means of the following analytical solution (van Genuchten and Alves,1982, Trapp and Brüggemann, 1989, Brüggemann et al. 1991) of the convection-dispersion-equation (see chap. 4.5).

$$C(x,t) = \begin{cases} PECini \times Term(x,t) & for \quad 0 < t \le t_E \\ PECini \times Term(x,t) - PECini \times Term(x,t-t_E) & for \quad t > t_E \end{cases} \tag{1}$$

with

$$Term(x,t) = 0.5 \times \exp\left[\frac{(v-u)\times x}{2\times D}\right] \times erfc\left[\frac{x-u\times t}{\sqrt{4Dt}}\right] + 0.5 \times \exp\left[\frac{(v+u)\times x}{2\times D}\right] \times erfc\left[\frac{x+u\times t}{\sqrt{4Dt}}\right]$$

x distance from the entry point [m]

t time [s]

t_E time that a water package needs to traverse a water body segment [s].

v flow velocity [m s-1]

D dispersion coefficient [m2 s-1]

$u = v \times \sqrt{1 + \frac{4kD}{v^2}}$ (If degradation rate k (s-1) = 0, u is identical to the flow velocity v).

$$erf\ (x) = \frac{2}{\sqrt{\pi}} \int_0^x e^{-t^2}\, dt$$

Here erfc is the "complementary error function", an ordinary function. For its definition, see Trapp and Matthies (1998) (chapter 7.3). The function is graphically presented in Figure 8-5.

with the properties

erf(-x) = - erf(x), 1-erf(x) = erfc(x), erf(0) = 0 and erf(∞) = 1.

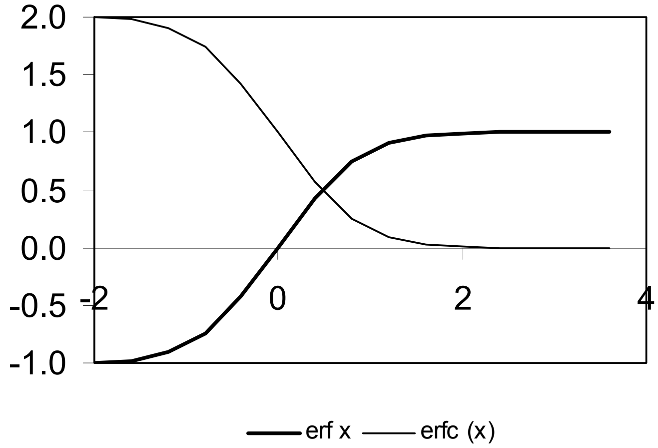

Figure 8-5: The error function erf (x) and the complementary error function erfc (x) = 1-erf(x).

This analytical solution of the convection-dispersion-equation is more realistic than the analytical solution presented in chap 4.6. The latter has a boundary condition leading to an entry of the total amount of the PPP to the water body segment at the point x_0 at time t_0. This means that the calculated concentration at the point x_0 at time t_0 (PEC_{ini}) is equal to infinity. This is and can never be real. In contrast to this, the solution of the equation presented here has a boundary condition resulting in a uniform distribution of the amount of PPP on the entry segment over the time t_E which is more realistic and corresponds to the assumption made in the standard scenario.

8.3.1.1 Calculation of relevant parameters at the middle point of each segment

The "$PEC_{dispers}$-Tool" is able to calculate various parameters that are relevant to assess the impact of the PPP depositions after drift at each middle point of a segment (PEC_{disp}, $PEC_{TWA(1h)}$, ToTh, RAC_{dyn} etc…). The calculation procedure of these parameters will be described in the following section.

PEC_{disp}

The PPP-concentration is calculated at the middle point of each segment at a given time. It is calculated as the sum of the different concentrations resulting from the different water packages that traverse the considered segment at the given time (see equation 2).

$$PECdisp(x,t) = \sum_{i=1}^{i=n} PECdispers_i(x,t) \quad , \quad i = 1,..,n \tag{2}$$

with

n = number of water packages that traverse the considered segment at the time t

x = middle point of the considered segment

PEC_{TWA} (PEC time weighted average) 1h

To calculate the $PEC_{TWA(1h)}$ for each segment the relevant time (i.e. the time from the first measurable concentration to the last measurable concentration at the middle point of the segment) is subdivided into equal intervals of 1 hour. For each hour the average concentration is calculated with the following equation:

$$PECtwa = \left(\sum_{i=0}^{i=n} \frac{\left(PECdisp(x, t_{i+1}) + PECdisp(x, t_i)\right)}{2} \times \Delta t \right) \bigg/ 3600 \qquad (3)$$

with:

t_i = sampling time, $t_{i+1} - t_i = \Delta t$ and $t_n - t_0 = 3600$ sec

Δt = sampling interval

x = middle point of the segment

The $PEC_{TWA(1h)}$ is the maximum out of the different PEC_{TWA} calculated by the mean of equation 3 over the relevant time.

The time over threshold (ToTh)

The time over threshold is the time period when the PEC_{disp} exceeds the RAC (static regulatory acceptable concentration: the threshold). It is calculated for the middle point of each segment by the mean of the following equation:

$$ToTh = n \times \Delta t \qquad (4)$$

with:

n = number of discrete times with $PECdisp > RAC$.

Δt = sampling interval

The dynamic regulatory acceptable concentration (RAC_{dyn})

The RAC_{dyn} is the RAC adapted to the ToTh and thus, suitable for short-time exposures (related to the exposure in the relevant toxicity test). It is calculated by the mean of the following equation:

$$RACdyn = RACsta \times 5.05 \times ToTh^{(0..348)} \qquad (5)$$

The randomized time of the application

As it is very difficult to asses the exact time for the application of the PPP by the farmers on the cultivated areas the $PEC_{dispers}$-Tool uses a random generator. Therefore, the application time slot is fixed for a certain period (typically 2 days, and it can be applied at each day from 8 a.m. to 6 p.m.), and for each segment of the considered water body the application time is generated using the mean of points in time produced by a random generator. Due to the fact, that we do not have any knowledge about the local ownership relations, we decided to use a randomised approach and did not consider potential auto-correlation of application events in nearby. As a kind of sensitivity analysis the integration of spatio-temporal autocorrelation effects could be realized in a follow-up project.

8.3.2 Model assumptions and preconditions

Underlying data: The water course network used here is the above mentioned corrected version of the ATKIS dataset. Land use information and corresponding geometries were derived from the ATKIS dataset by applying a 10 m buffer zone around the relevant surface water bodies. The definition of relevant water bodies was taken over from the static model.

Calculation of initial spray drift entries: When developing the dynamic modeling approach the methods and modeling results described in chapter 4 (PEC_{ini}) were not yet applicable and hence could not be implemented. Therefore, a basic GIS-based algorithm adapting the concepts of the previously developed static model for the calculation of drift entries was modified and employed. The calculation of drift entries (PEC_{ini}) follows the functions (90. percentile) derived from spray drift experiments carried out at JKI-Germany (www.jki.bund.de/fileadmin/dam_uploads/_AT/.../Abdrifteckwerte_xls.xls).

Furthermore the calculation of PEC_{ini} in the dynamic model is based on real values for hydrological parameters obtained in several field campaigns (Trapp et al. 2008b). So the assumptions made here are based on "real world"-observations and measurements in the study area. This is in contrast to the static modeling approach which assumed a simplified concept with standardised values for width, length and depth of the water bodies.

Two water bodies were selected in the hop region Hallertau.

In a first step all water bodies used for the simulations were split up into 25 m segments. Secondly an application rate of 1000 g/ha was assumed. These two steps are in accordance with the standard approach of GeoRisk.

Concerning the origin of drift entries the main assumption is that the highest deposition value out of eight wind directions is always taken into account as PEC_{ini} (DepMax). This means that regarding each segment of the stream step by step, drift entries always come (i.e. the wind always blows) from the nearest application field.

Realistic application patterns were implemented replacing those assumed in prior GIS-based modeling approaches. Therefore a number of 25 Monte-Carlo simulations were carried out for the selected water bodies within an application time slot of 20 hours over 2 days (application time 8 a.m to 6 p.m). In this time frame randomised depositions from the application areas into the surface water body segments (20 hours) were simulated.

Simulations were carried out only for the main branch of a stream network with stationary (though realistic) hydrological boundary conditions concerning depth, width and flow velocity of the water body. Tributary streams have not been taken into account and therefore did not influence the results. Their influence can vary at a wide range depending mainly on the structure of their neighbouring areas, especially the possibility of pesticide entries from agricultural land (here: hops). In the case at hand the tributary streams do pass hop fields and are therefore exposed to spray drift. It may be expected that, if taken into account in the simulations, these entries might have lead to higher results concerning the observed parameter time over threshold also in the main branch.

Abbreviations

RAC: regulatory acceptable concentration

RAC_{sta}: RAC for static water bodies, corresponds to the RAC used in the current approach (Generic RAC for hop = 6 µg/L, resulting from 1 kg/ha and 20 m distance)

ToTh: time over threshold (RAC_{sta})

RAC_{dyn}: dynamic RAC following the approach of short-time exposure1 (*$RAC_{dyn} = RAC_{sta} * 5.05 * ToTh^{(-0.348)}$*): RAC adjusted to the ToTh

RAC_{dyn}To: the adapted RACdyn to the investigated „$_{TWA(1h)}$"

TWA(1h): time-weighted average about 1 hour

PEC_{TWA}: maximum of the different average concentrations about 1 h within 20 h

$TWA_{\_SA}$: scenario with simultaneous application patterns for all segments

Twa_{Max}: Maximum of the „$PEC_{TWA(1h)}$"

Twa_{Min}: Minimum of the „$PEC_{TWA(1h)}$"

Twa_{90P}: 90^{th} percentile of the „$PEC_{TWA(1h)}$"

Twa_{50P}: 50^{th} percentile of the „$PEC_{TWA(1h)}$"

(25 simulations with randomised application patterns)

In summary:

While the risk assessment assuming lentic water bodies is based on the comparison of the PECini with the RAC, the local exposure pattern predicted by the dynamic model is summarized to the maximum TWA over 1 h ($PEC_{TWA(1h)}$ and the total duration when the PEC is above the RAC (ToTh).

Because the local PEC is depending on the variable timing and magnitude of the PPP entries upstream, Monte-Carlo distributions provide a set of possible exposure pattern for each segment from which different $PEC_{TWA(1h)}$ can be extracted (e.g. minimum, maximum, median).

The exposure duration (as ToTh) is considered by calculation of a RAC_{dyn} to consider that effect thresholds are higher it the exposure duration is shorter.

8.4 Results of 25 simulations with randomised application patterns for the Lauterbach and the Haunsbach

8.4.1 Geographical context of the Haunsbach and the Lauterbach

Both streams analysed here, the Haunsbach nearby Meilenhofen and the Lauterbach, are situated in the centre of the hop growing region Hallertau.

The Haunsbach is draining into the river Abens, the Lauterbach into the river Ilm.

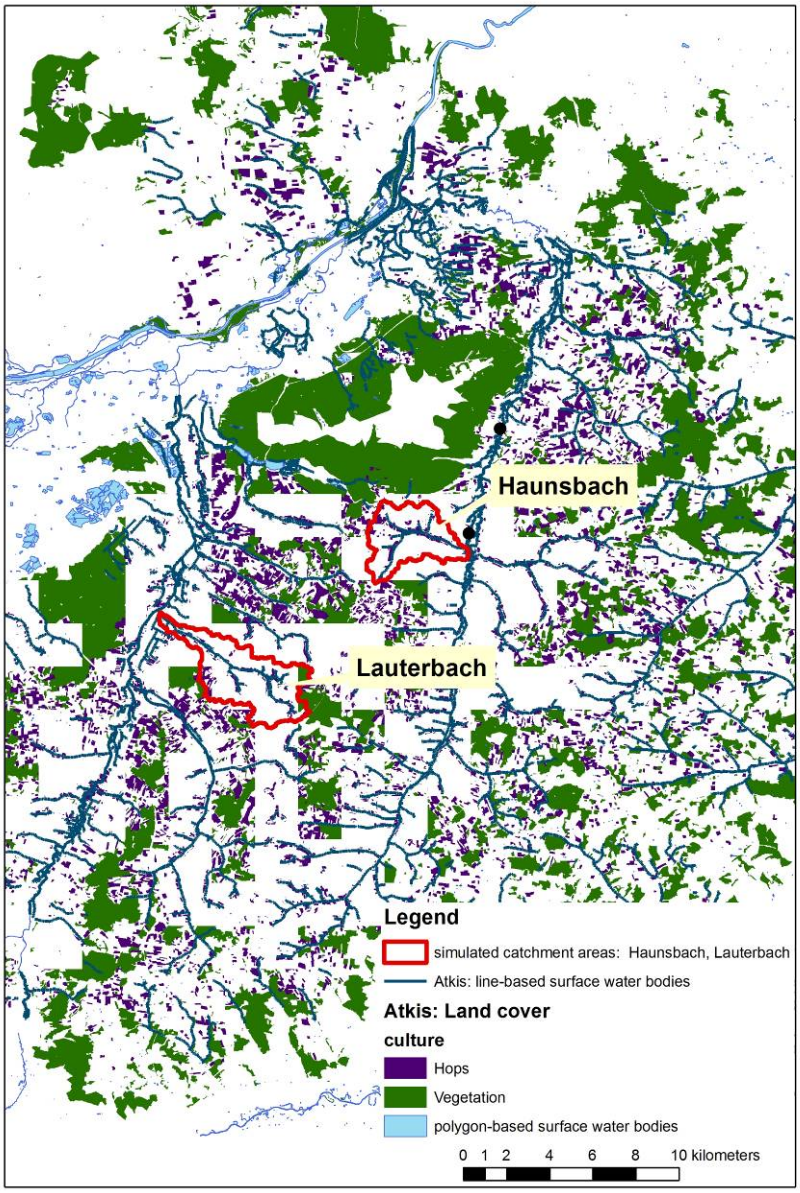

Figure 8-6: Overview of the catchments of the both investigated water bodies in the hop growing region Hallertau

8.4.2 Geographical context of the Haunsbach

Nearly 6 km stream length of the Haunsbach were used for calculating the spatio-temporal dynamic PEC. The simulation only integrated the main branch of the water body. The calculation ended at the connection of the Haunsbach to the receiving water body Abens.

The hydrological parameters were surveyed in three ground truthing campaigns. Methodology and the results are documented in detail in a separate report (Trapp et al. 2008B and 2009).

The mean flow velocity of the Haunsbach is 0.22 m/s, the mean water depth is 0.10 m and the mean water body width is 0.66 m. Within a 150 m buffer zone around the water body the hop area amounts to 69 ha.

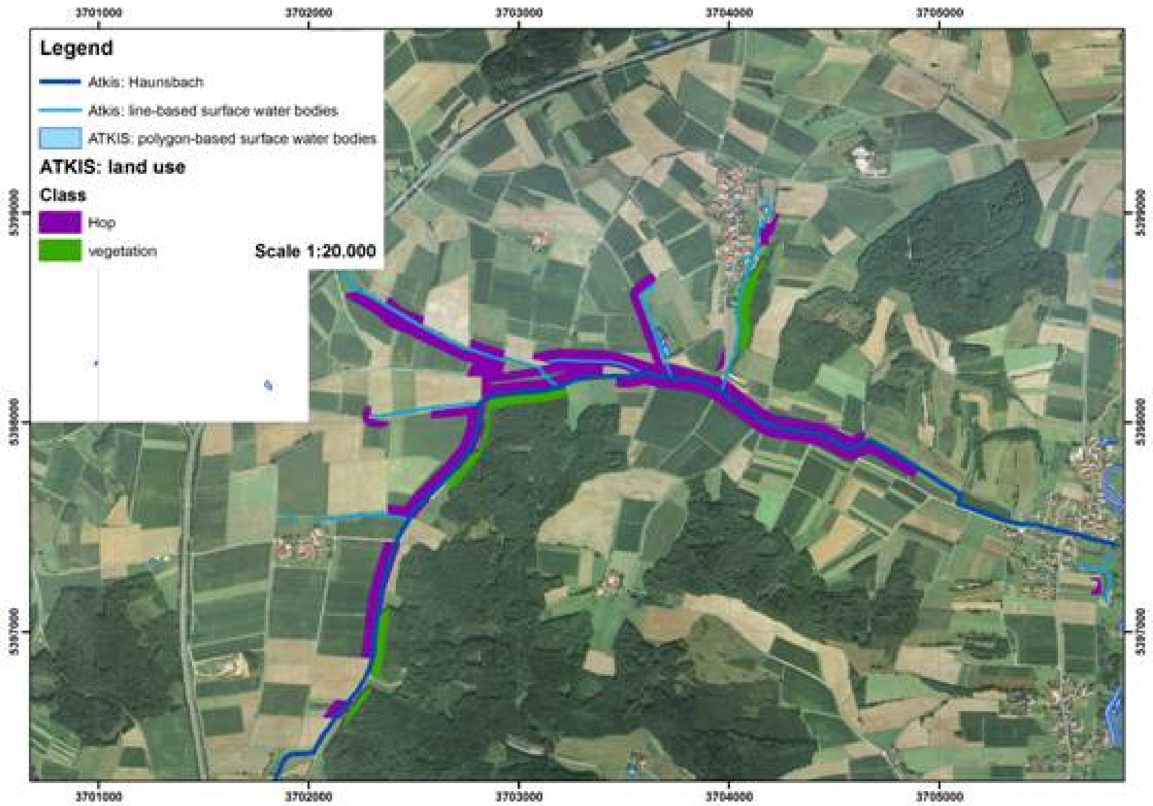

Figure 8-7: Detailed map of the Haunsbach

The next figures show pictures from ground truthing of the Haunsbach.


Figure 8-8: Pictures from ground truthing of the Haunsbach: June 2008 (left), March 2009 (right)

8.4.3 Geographical context of the Lauterbach

Nearly 8.6 km stream length of the Lauterbach were used for calculating the spatio-temporal dynamic PEC. The simulation only integrated the main branch of the water body. The calculation ended at the confluence of the Haunsbach to the receiving water body Ilm.

The hydrological parameters were surveyed in three ground truthing campaigns.

Methodology and results are documented in detail in a separated report (Tapp et al. 2008B and 2009).

The mean flow velocity of the Haunsbach is 0.25 m/s, the mean water depth is 0.11 m, and the mean water body width is 0.52 m.

Within a 150 m buffer zone around the water body the hop area amounts to 58 ha.

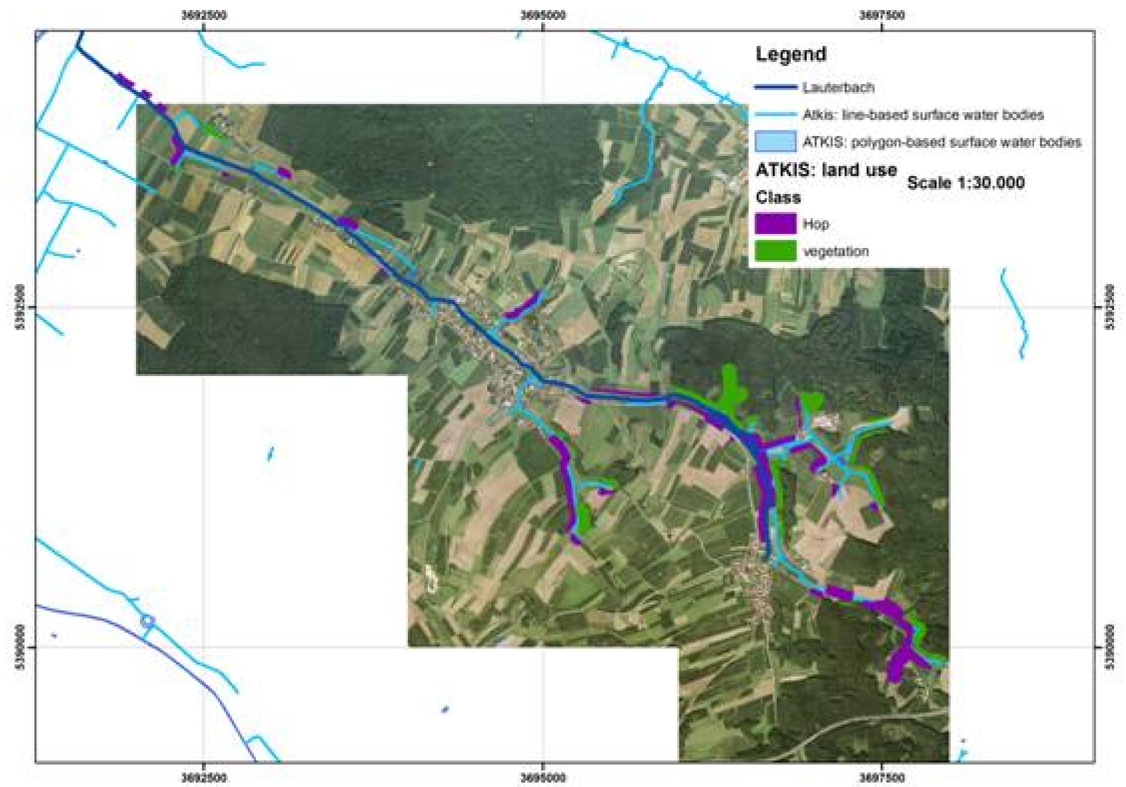

Figure 8-9: Detailed map of the Lauterbach

The next figures show pictures from ground truthing of the Lauterbach.

Figure 8-10: Pictures from ground truthing of the Lauterbach June 2008 (left), March 2009 (right)

8.5 Results of the dynamic modelling for the Haunsbach

In this chapter the results of the 25 simulations with randomised application patterns for the Haunsbach will be described (results see Figure 8-11).

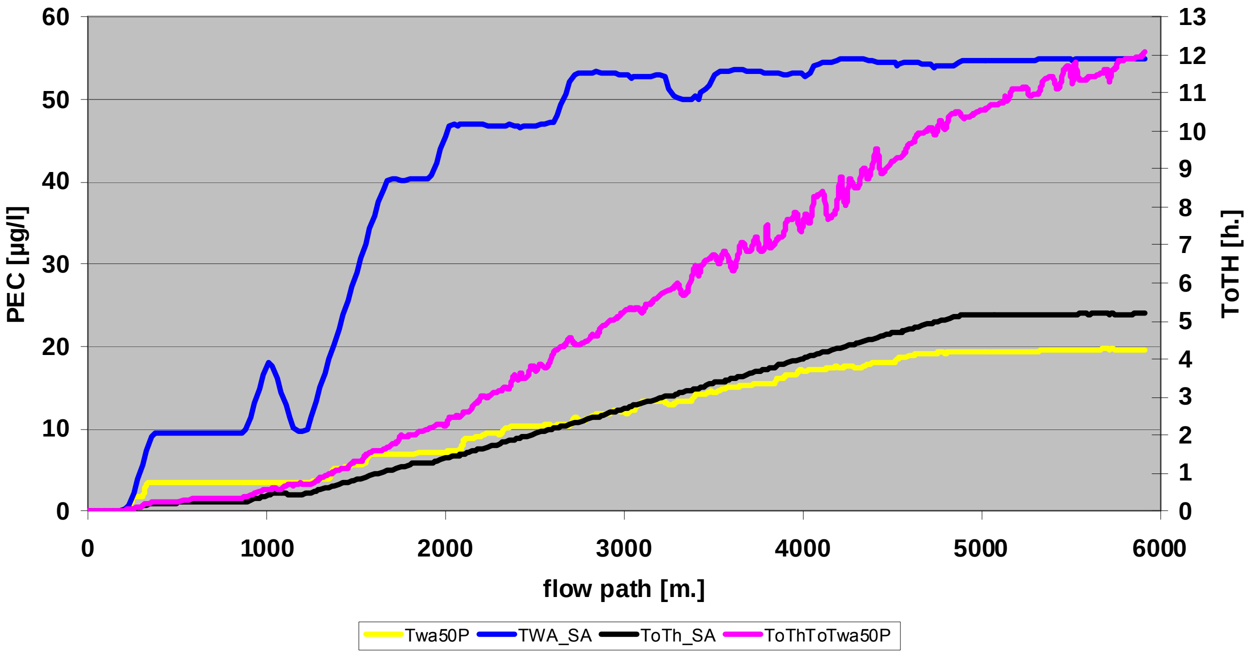

Figure 8-11: Results for a simultaneous application and the 50. percentile of 25 simulations with randomized application patterns for the Haunsbach

It can be seen clearly that using the dynamic model only short-time exposure occurs, maximally 5 hours above threshold.

Another point is that the simultaneous application pattern is the worst case application pattern.

The consortium therefore discussed which PEC percentile of the Monte-Carlo simulations of application patterns would be the most relevant one. As the exact application pattern is unknown, an equal temporal distribution of the applications was chosen. Because the estimation of the drift entries into each upstream segment is based on the selection of the maximum value calculated for all wind directions (and thus, it is assumed that the wind always blows from the nearest application area directly to the next water body segment along the whole simulated stream), the choice of a higher percentile than the 50. centile for the local PEC distribution considering the upstream application pattern would result in an accumulation of worst case assumptions in this context. Thus, the median of the local $PEC_{TWA(1h)}$ values was used.

The next figure compares the different percentiles of the $PEC_{TWA(1h)}$.

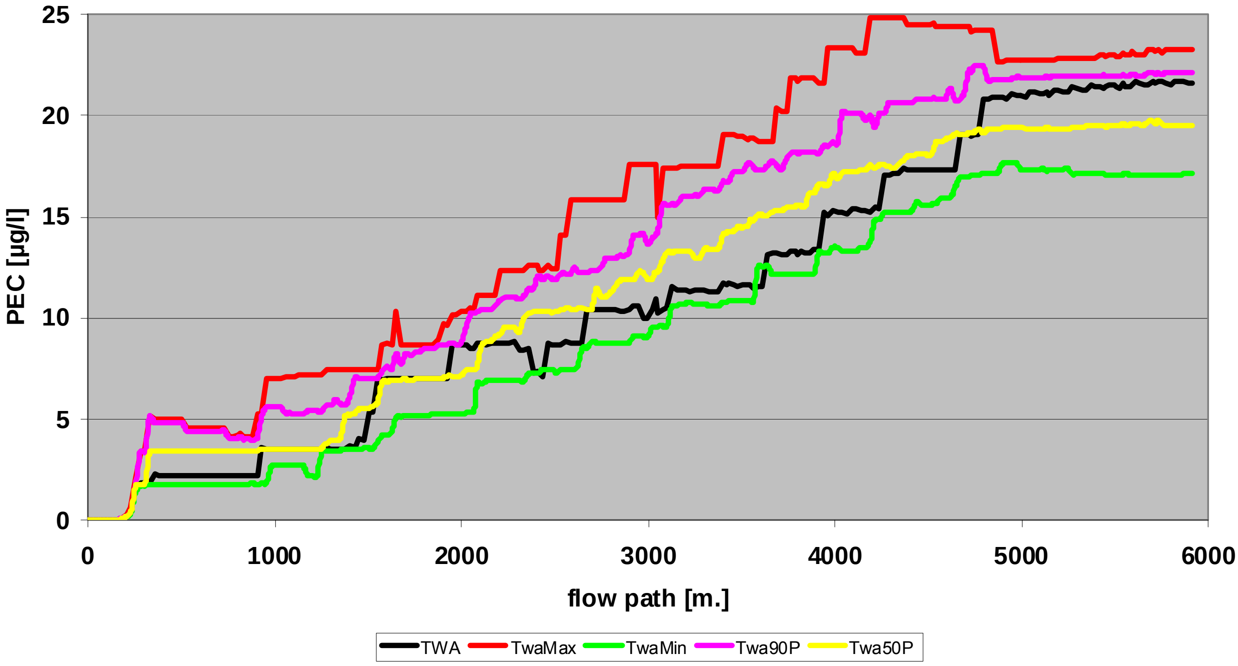

Figure 8-12: Results of the PEC_{TWA} of all percentiles of the 25 simulations with randomized application patterns for the Haunsbach

The range between the PEC_TwaMax 1h and the PEC_TwaMin 1h is not as evident as the choice of the application pattern (all applications simultaneously or within 2 days).

8.6 Results of the dynamic modelling for the Lauterbach

In this chapter the results of the 25 simulations with randomised application patterns for the Lauterbach will be described (results see Figure 6-13).

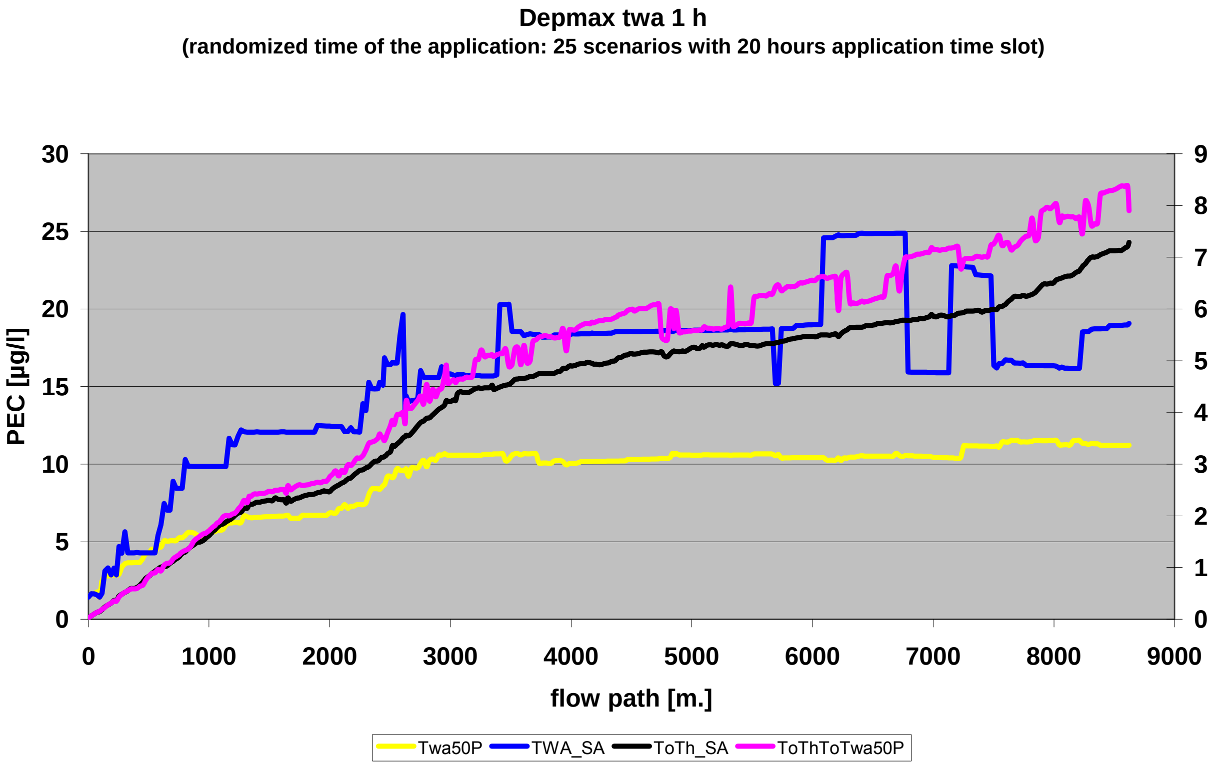

Figure 8-13: Results for a simultaneous application and the 50. percentile of 25 simulations with randomized application patterns for the Lauterbach

Here a different pattern of the simultaneous application (SA) occurs. Due to the fact that there are segments without hops nearer than 150 m one can see depressions of the curve of the PEC_TWA_SA.

Randomising the application this effect does not have such evidence.

The simultaneous application remains the worst case pattern.

The flow path is 8.5 km, so it takes more than 9 hours to flow from start to end, and therefore the maximum time over threshold is about 7 hours.

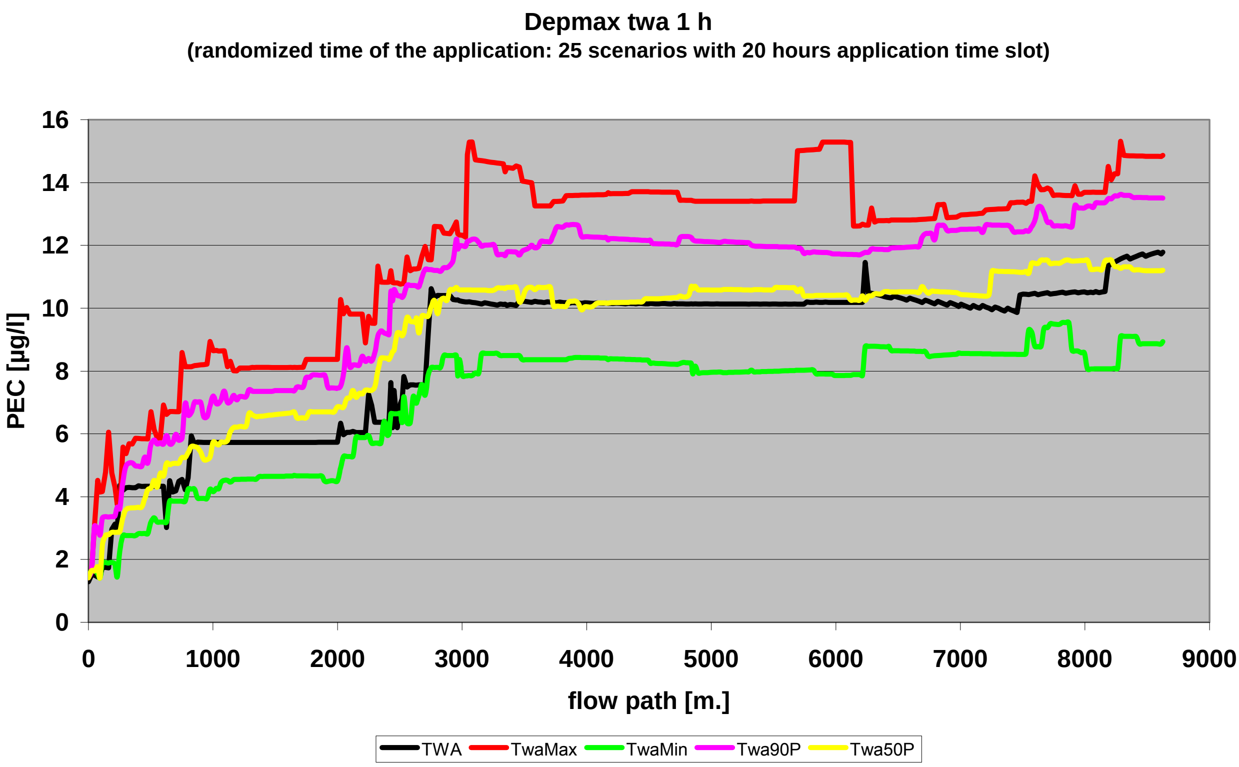

Figure 8-14: Results of the PEC_{TWA} of all percentiles of the 25 simulations with randomized application patterns for the Lauterbach

Here the results are similar to the results of simulating the Haunsbach.

The range between the PEC_TwaMax 1h and the PEC_TwaMin 1h is not as evident as the choice of the application pattern (all applications simultaneously or within 2 days).

The next figure visualizes the results of the dynamic modelling with a simultaneous application pattern as a map.

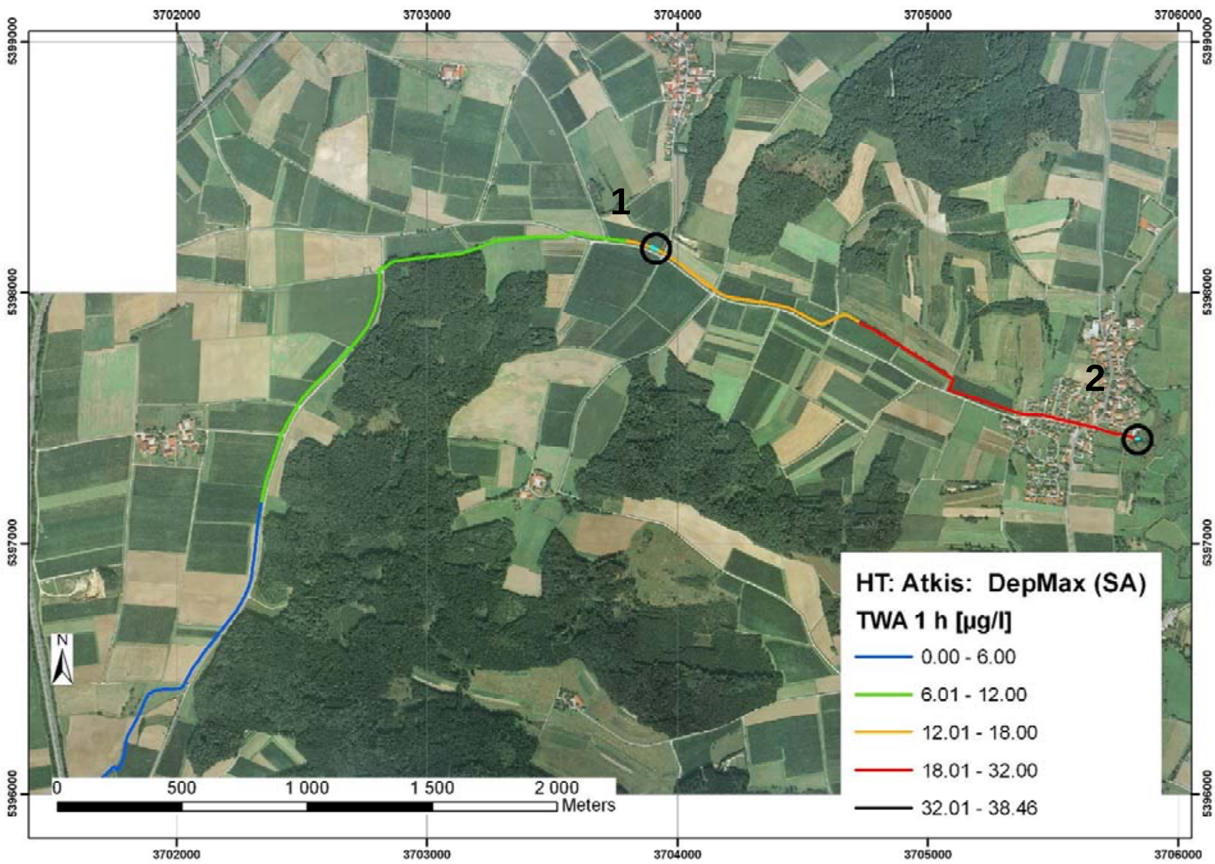

Figure 8-15: Results of the PECTWA of all percentiles of the 25 simulations with randomized application patterns for the Lauterbach

One randomized temporal distribution of the dynamic PEC for the simultaneous application pattern is shown for the 2 marked segments (black bolds) in the following figures.

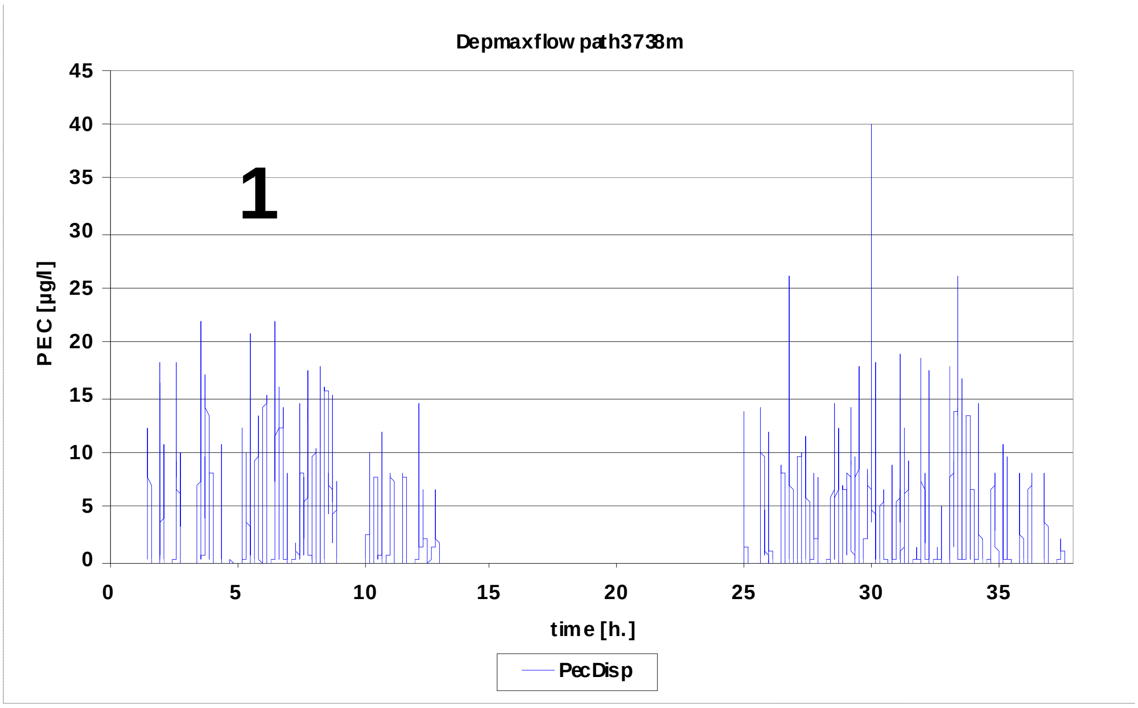

Figure 8-16: One random temporal distribution of the dynamic PEC for the simultaneous application pattern after a flow path length of 3738 m

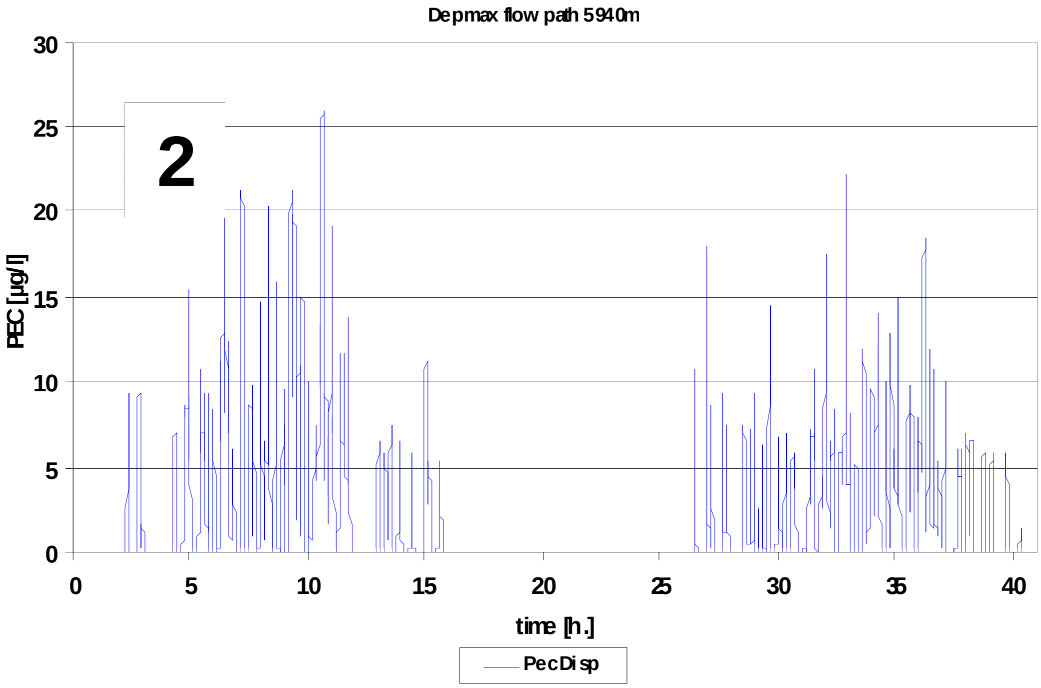

Figure 8-17: One random temporal distribution of the dynamic PEC for the simultaneous application pattern after a flow path length of 5940 m

It can be seen that the patterns are similar, but not identical. For each of the 25 simulations and each segment these PEC-values can be visualized and each single application pattern would result in different PEC-values for each segment.

8.7 Consequences for the framework of PPP authorisation

In the static model, the threshold for defining segments of risk (segments with PEC_{ini} > RAC) is the comparison between the initial PEC and the RAC.

Segments with risk in static water bodies: **PEC_{ini} > RAC**

When implementing a dynamic modelling in flowing waters, a new threshold has to be used.

Segments with risk in flowing waters: **$PEC_{TWA(1h)}$ > RAC_{dyn}**

(following the approach to considere short-time exposure in the effect assessment)

Therefore, this new threshold was used to calculate the segments of risk of the two investigated brooks Haunsbach and Lauterbach.

The calculation of these segments of risk has not yet included the concept of the moving window (hotspot criterion) and also the calculation of effect levels per segment (using generic or substance related dose response functions) or consideration of multiple applications were not implemented yet because the focus here was on the modelling of the dynamics of the exposure. However, intergration hotspot criteria with flexible acceptable effects levels as proposed in chapter 6 and dose-response curves is no major technical problem if the approach should be implemented on a larger scale. In the dynamic model the segments with exposure and effects are not directly adjacent. Therefore this mowing window approach had to be adapted. This was not realised.

8.7.1 Results of hotspot calculation based on the dynamic modelling of the Haunsbach

The next table shows the results of the dynamic modelling in the context of calculating the hotspots following the preliminary proposal of the generic hotspot criteria (see chapter 6).

Table 8-1: Results of the 25 scenarios based simulations for the Haunsbach with randomised application patterns

	Flow path [m]	PEC > RAC_{sta} [m]	$PEC_{TWA(1h)}$ > RAC_{dyn} [m]	Reduction* [%]
Twa_{Max}	5932	3918	3009	23
Twa_{90P}	5932	3918	2632	33
Twa_{50P}	5932	3918	2268	42
Twa_{Min}	5932	3918	1792	54

*Compared to the static PEC_{ini} calculation with realistic hydrological parameters from ground truthing

In the static model a length of 3918 m out of 5932 m all in all (66 %) is defined as segments with risk by using the 50th percentile of the 25 simulations with realistic hydrological parameters. These realistic hydrological parameters were derived from three ground truthing campaigns. From the upper reaches to the lower course of the stream randomised locations were chosen to measure water depths and flow velocities. The mean value of all measures was used as input parameter for the calculation of the PEC_{ini} static as well as for the PEC_{ini} dynamic (s. also Chapter 4.6.7 and Trapp et al. (2009) as well as Trapp et al. (2008B).

The next figure points out the differences of the PEC_{ini} between the static and the dynamic model taking into account the different RAC in both approaches, the usual RAC and the RAC_{dyn} considering pulse exposure.

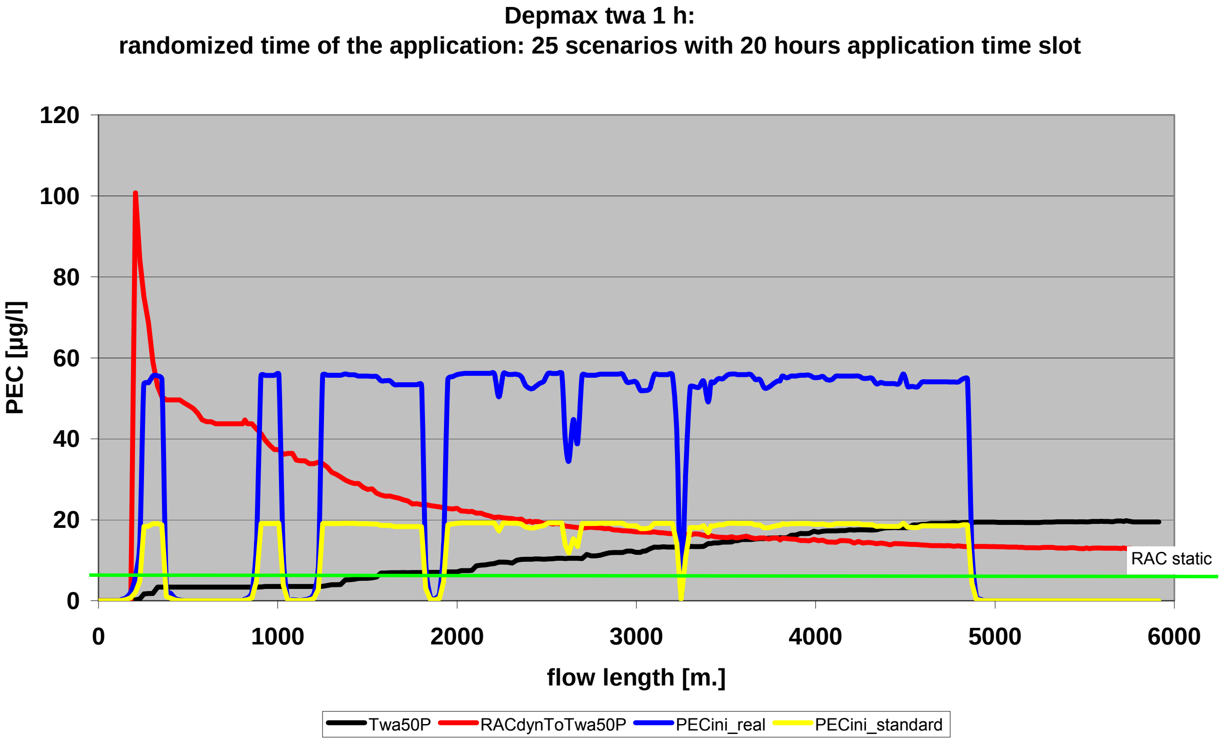

Figure 8-18: Differences between the PEC_{ini} static with the RAC static and the PEC_{ini} dynamic with the RAC dynamic for the 25 randomized application patterns for the Haunsbach

The yellow line represents the results for the PEC_{ini} static with the standard water volume of 300 l, the green line represents the RAC static of 6 µg/l. The blue line represents the PEC_{ini} static with realistic hydrological parameters derived from ground truthing as described above (a detailed description of the methodology is given in Trapp et al. (2008B) and Trapp et al. (2009)). The black line represents the results of the dynamic approach, here the $_{TWA(1h)}$ of the 50 percentile as described in Ch. 8.5, and the red line represents the RAC dynamic following the concept of short time exposure as described in detail in Ch. 6.2.8.4.

As it can be clearly seen the PEC_{ini} static based on the standard water volume of 300 l in some cases leads to an underestimation of the initial PEC concentration in smaller water bodies. The RAC static and the RAC dynamic differ.

Conclusions

These results can lead to different assumptions:

First of all, the standard approach using a standardised ditch with 300 L water volume for calculating the $PEC_{initial}$ static is not always a conservative assumption due to different hydrological parameters in reality, especially different water depths and water volumes.

When integrating more realism into the PEC-calculation a dynamic modelling is necessary.

In the dynamic model a length of 2268 m out of 5932 m all in all (38 %) is defined as segments with PEC_{ini} > RAC by using the 50th percentile of the 25 simulations with realistic hydrological parameters. This corresponds to a reduction of 42 % between the static model and the dynamic model.

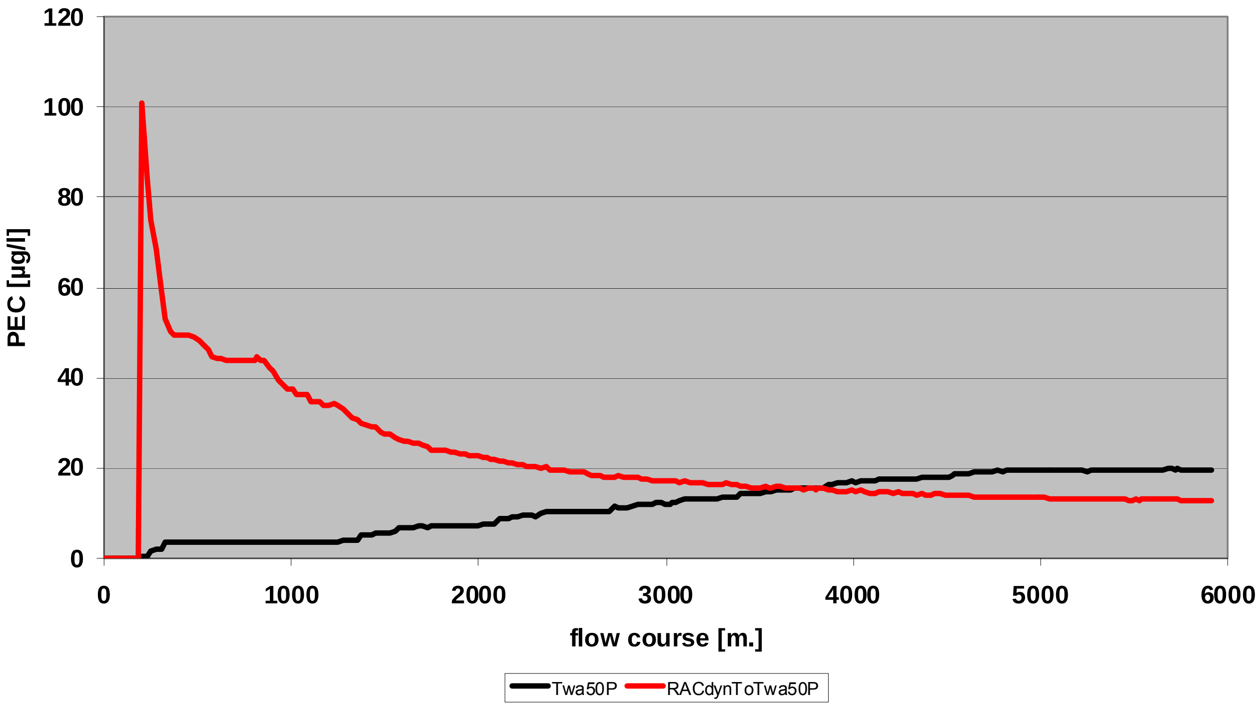

Figure 8-19: Segments with PEC > RAC regarding the $PEC_{TWA\_50th\ centile}$ for the Haunsbach by using the dynamic model

This figure shows the graphical presentation of the results of the dynamic modelling with the dynamic RAC.

8.7.2 Results of hotspot calculation based on the dynamic modelling of the Lauterbach

The next table shows the results of the dynamic modelling in the context of calculating segments with PEC > RAC following the assumption of the German Environmental Protection Agency (Umweltbundesamt, UBA).

Table 8-2: Results of the 25 scenarios based simulations for the Lauterbach with randomized application patterns

	Flow path [m]	PEC > RAC_{sta} [m]	$PEC_{TWA(1h)}$ > RAC_{dyn} [m]	Reduction* [%]
Twa_{Max}	8637	2924	350	88
Twa_{90P}	8637	2924	0	100
Twa_{50P}	8637	2924	0	100
Twa_{Min}	8637	2924	0	100

*Compared to the static PEC_{ini} calculation with realistic hydrological parameters from ground truthing

In the dynamic model using the 50th percentile of the 25 simulations no segments with PEC > RAC remain for the Lauterbach.

This means that implementing the dynamic modelling of PEC segments with PEC > RAC exceedings of the threshold are depending from application pattern, water depth and flow velocity. In the case study of the Lauterbach, only the application pattern with the maximum $PEC_{TWA(1h)}$ of the 25 randomised simulations leads to segments with PEC > RAC. All other application patterns do not lead to exceedings and therefore no hotspots would remain.

The next figure shows the graphical presentation of these results.

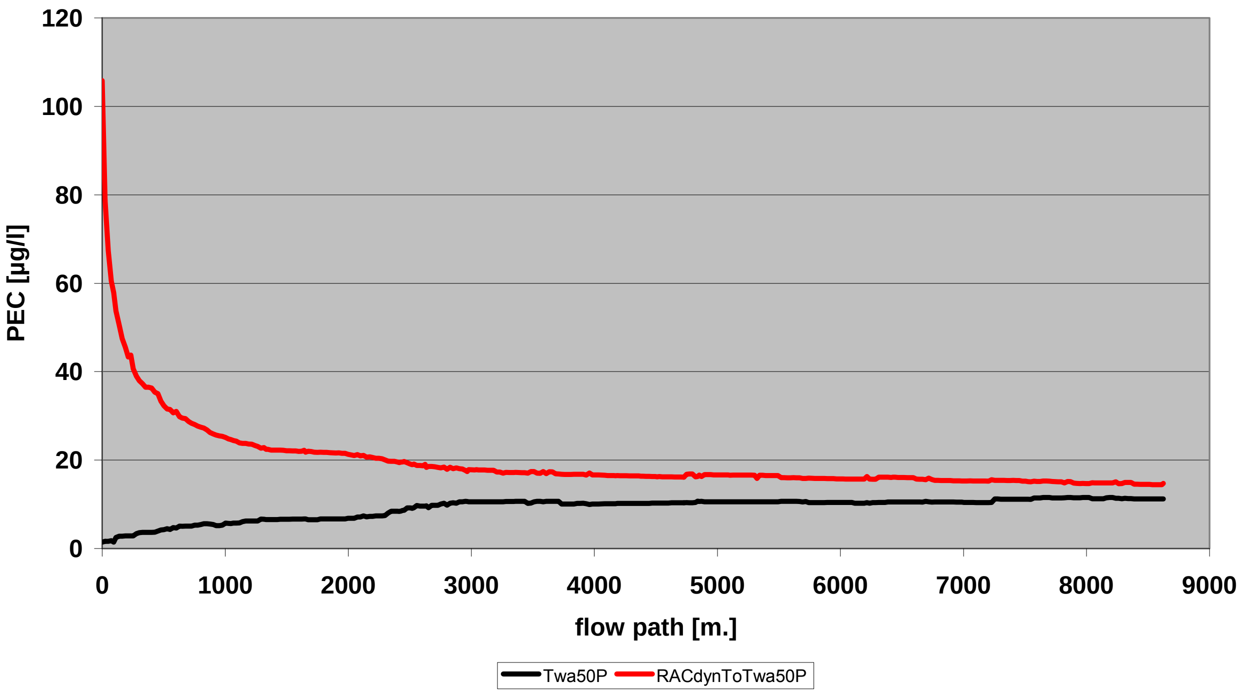

Figure 8-20: Segments with PEC > RAC regarding the $PEC_{TWA\_50th\ centile}$ for the Lauterbach by using the dynamic model.

8.8 Analysis of representativeness of the Haunsbach and the Lauterbach

8.8.1 Index of Exposure

Interpreting the results of the two investigated streams Haunsbach and Lauterbach the important questions are the reliability and the representativeness of the results.

Therefore an analysis of representativeness of the two selected water streams Haunsbach and Lauterbach compared to all streams in the hop region Hallertau was conducted by using results of the project "GeoPERA" (Wagner et al. 2007, this study report was provided to the UBA and the consortium).

In this project the so-called index of exposure was calculated for whole Germany based on the ATKIS-dataset. For this purpose a grid with 4 km^2 (2 x 2 km) was drawn for all hop growing regions in Germany. This grid was oriented on the gridding of the aerial images of the Geodesy and Geodata Agency of Rhineland-Palatinate geo-referenced in Gauss-Krueger Zones.

In a first step all grids containing relevant streams were selected (line-based streams form the ATKIS-dataset with the Objektart 5101 and 5103 and the attribute OFL 1100 situated totally within a buffer zone of 150 m around the hop fields).

In a second step these relevant streams were divided into segments of 25 m and the mean PEC of 8 wind directions was calculated. For each of these grids all mean values were summarized. This sum of concentrations describes the range of the potential entry of plant protection products in the streams of each grid.

Following this approach the maximum per grid of the summarized potential entries in Germany is nearly 7400 µg/l, the 90th percentile is nearly 1240 µg/l.

Table 8-3: German-wide results of the summarized deposition per grid [µg/l]

Percentile:	90	95	100
German-wide:	1237	1867	7395

The following figures are derived from the project GeoPERA (Wagner et al. 2007).

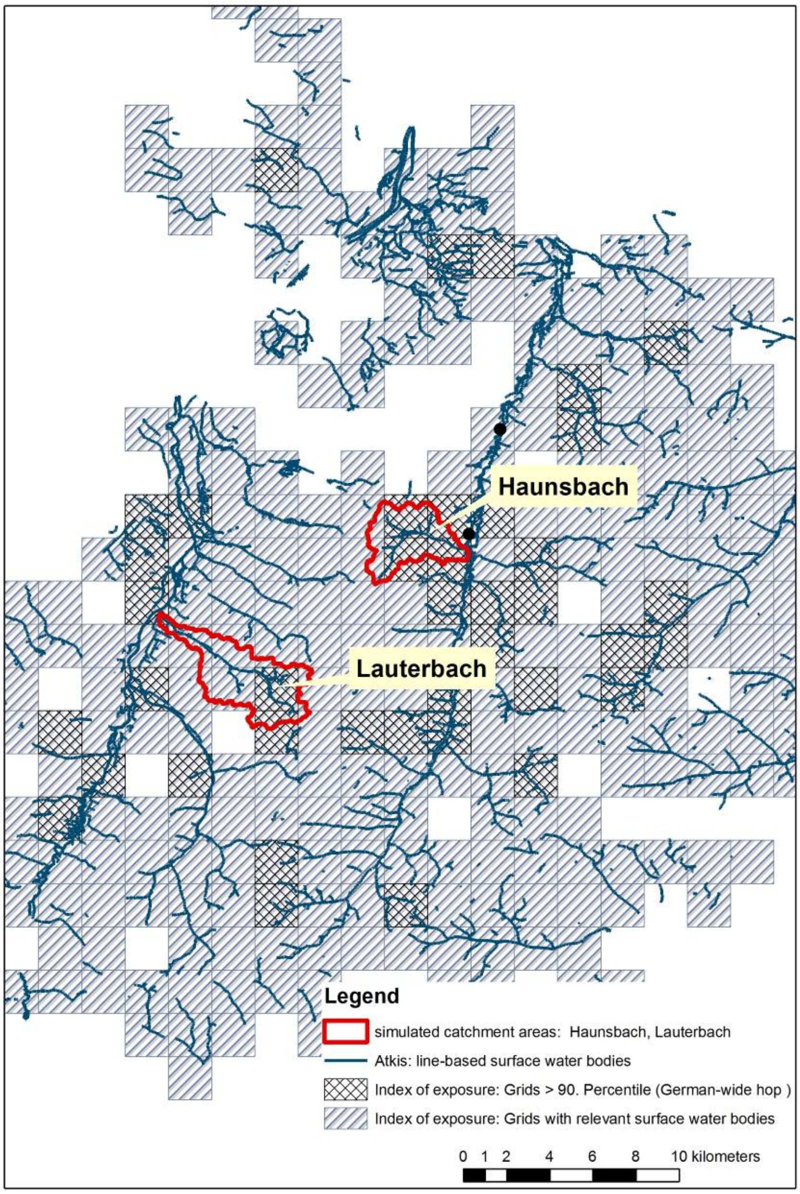

Figure 8-21: Map of indices of exposure of the hop growing region Hallertau

Presented are all grids containing relevant water bodies as well as grids with an index of exposure up to the 90th percentile compared with the German-wide distribution.

The catchment of the Haunsbach intersects 4 adjacent grids with an index of exposure up to the 90th percentile, the catchment of the Lauterbach crosses 2 of these grids.

Table 8-4: Values of the index of exposure of the grinds of the selected catchments (std: standard deviation)

	Min	max	mean	sum	std
Hallertau	-	-	419.67	-	786.85
Haunsbach:	1107	5600	3242	16212	1326.04
Lauterbach	455	5724	1768	12375	1352.00

Therefore the maxima per grid situated partially or totally within the catchments are up to the German-wide 99th percentile.

When regarding the mean values of the indices of exposure per grid situated partially or totally within the catchments, the mean values of the catchment Haunsbach are up to the 98th percentile, the mean values of the catchment Lauterbach up to the 93th percentile.

The following figure shows the summarized curve of the German-wide index of exposure as well as the results concerning the two catchments.

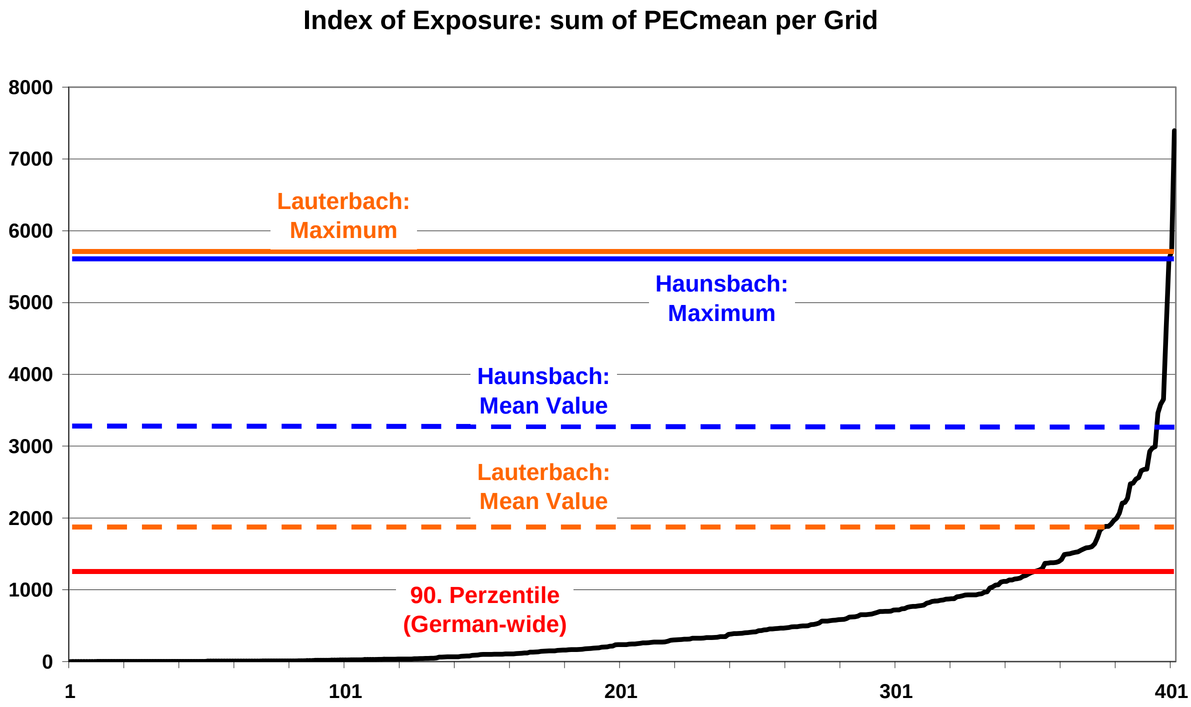

Figure 8-22: Summarized curve of the German-wide index of exposure as well as the results concerning the two catchments (only grids with sum > 1 are shown)

Following the results of the analysis of representatives concerning the index of exposure both surface water bodies can be described as a conservative assumption in a German-wide comparison. Especially the Haunsbach constitutes a so-called worst case with a percentile higher than the 95th percentile of the German-wide distribution.

8.9 Representativity of the hydrological parameters

During several ground truthing actions in 2008 and 2009 the bulk of relevant surface water bodies classified as potentially containing segments with PEC > RAC (RS) was mapped in detail concerning their hydrological parameters (Trapp et al. 2008b, 2009).

Table 8-5: Hydrological parameters of selected water courses in the hop growing region Hallertau (std: standard deviation)

	with [m]				depth [m]				flow velocity [m/s]			
	mean	min	max	sd	mean	min	max	SD	mean	min	max	std
RS mapping 2008 (n=39):	0.51	0.05	1.75	0.34	010	0.02	0.35	0.08	-	-	-	-
RS mapping 2009 (n=61)[49]:	0.48	0.15	1.35	0.30	0.10	0.03	0.30	0.05	0.23	0.0	1.00	0.18
Haunsbach	0.66	-	-	-	0.10	-	-	-	0.22	-	-	-
Lauterbach	0.52	-	-	-	0.11	-	-	-	0.25	-	-	-

Therefore, comparing all brooks in the hop growing region Hallertau measured by ground truthing in detail the hydrological parameters of the Haunsbach and Lauterbach can be regarded as being typical for this region.

For a classification of the representativeness regarding the catchment level a comparison was conducted with 5 additional catchments with different structures of land use pattern and different length of the water courses.

The length of the brooks within the catchments of Haunsbach and Lauterbach are in the mean or lower, but the percentage of the maximal initial PEC per catchment is very high. This underlines the higher-than-average potential risk of exposure of the both selected water courses. Even the high percentage of hop fields underlines this.

[49] not published

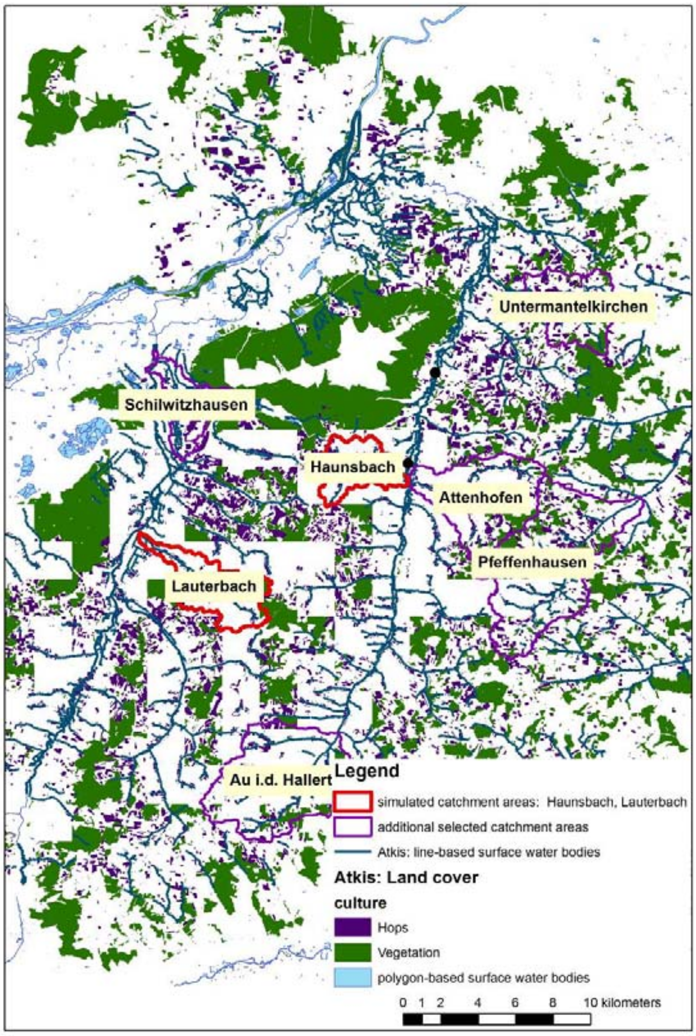

Figure 8-23: General map of the selected catchments

Table 8-6: Hydrological parameters and land use patterns of selected catchments in the hop growing region Hallertau

	Length of relevant water courses [km]	Catchment area [ha]	Area of hop fields [ha] (within the 150 m buffer)	Area of the 150 m buffer zone [ha]	Ratio of hop area [%]	Sum PEC_{ini} static Max [%]
Au i. d. Hallertau	29.22	3508	185.3	948.2	19.5	100.0
Haunsbach	10.0	1040	106.0	281.2	37.7	95.6
Lauterbach	16.4	1503	96.3	432.7	22.3	90.8
Pfeffenhausen	23.0	4078	152.4	727.6	20.9	87.4
Attenhofen	18.8	2413	128.1	593.6	21.6	75.9
Schilwitzhausen	10.6	964	48.4	332.9	14.5	59.5
Untermantelkirchen	10.1	1550	55.8	309.5	18.0	42.3

8.10 German-wide extrapolation of the results

An extrapolation to calculate the German-wide ratio of segments with PEC > RAC was conducted using the index of exposure.

The GeoPERA-project used the ATKIS DLM2 dataset for calculating this German-wide index of exposure based on the definition of relevant water bodies (totally within a 150 m buffer zone around the hop fields) as defined in the GeoRisk-project (see chapter 5.3.3) as well as in earlier projects, too. In chapter 7.2 the results of the hotspot analysis using the static exposure model are given (see also Table 7.1). For hops 1019 km relevant water bodies were extracted, containing 146 km RS (risk segments) and 141 MS (management segments).

Regarding the Lauterbach, 7 % of all surface water bodies in German hop growing regions have a higher index of exposure. This leads to 70 km of relevant surface water bodies.

Integrating the concepts of a dynamic water body system based on real hydrological parameters a ratio of 12 % segments with PEC > RAC in comparison to the static model of water bodies was calculated regarding the maximum $PCE_{TWA(1h)}$ of 25 randomised application patterns. **This leads to 8.4 km risk segments (PEC > RAC) for all relevant surface water bodies in Germany.** Regarding the 90^{th} percentile of the $PEC_{TWA(1h)}$ of the 25 randomized simulations of the application patterns as well as the 50^{th} percentile no risk segments (PEC > RAC) were calculated.

Regarding the Haunsbach less than 2 % of all relevant surface water bodies in German hop growing regions have a higher index of exposure. This leads to 20 km relevant surface water bodies. Integrating the concepts of a dynamic water body system based on real hydrological parameters a ratio of 77 % risk segments (PEC > RAC) in comparison to the static model of water bodies was calculated regarding the maximum $PEC_{TWA(1h)}$ of 25 randomised application patterns**. This leads to 15.4 km risk segments (PEC > RAC) for all relevant surface water bodies in the German hops growing regions**. Regarding the 90^{th} percentile 13.4 km risk segments (PEC > RAC) were calculated, regarding the 50^{th} percentile of the $PEC_{TWA(1h)}$ of the 25 randomized simulations of the application patterns 11.6 km risk segments (PEC > RAC) remain.

Overall nearly 10 % of the static RS remain RS when implementing the dynamic model taking into account the real hydrological conditions of the surface water bodies.

Transferred to all permanent crop regions in Germany (except the “Altes Land”) it can be carefully assumed that under the prerequisites made (see assumptions listed in 8.3.2) for calculations here, the expected management segments should be around 200 km for whole Germany.

8.11 Analysis of sensitivity of selected input parameters

8.11.1 Effects of dispersion

Analysing possible effects of dispersion based on the implementation of the dynamic model different flow velocities were simulated. Due to the fact that the hydrological parameters have to be constant in our model, only the main part of the water courses was used in the simulation. All other related parameters remain constant so that the influence of different flow velocities of the TWA could be estimated.

In the following figure the results of flow velocities differing from 0.1 m/s to 0.4 m/s are pointed out (PECDisp_FV1, _FV2, _FV3, _FV4) for the segment next to the junction with the river Abens after a flow length of 5940 m.

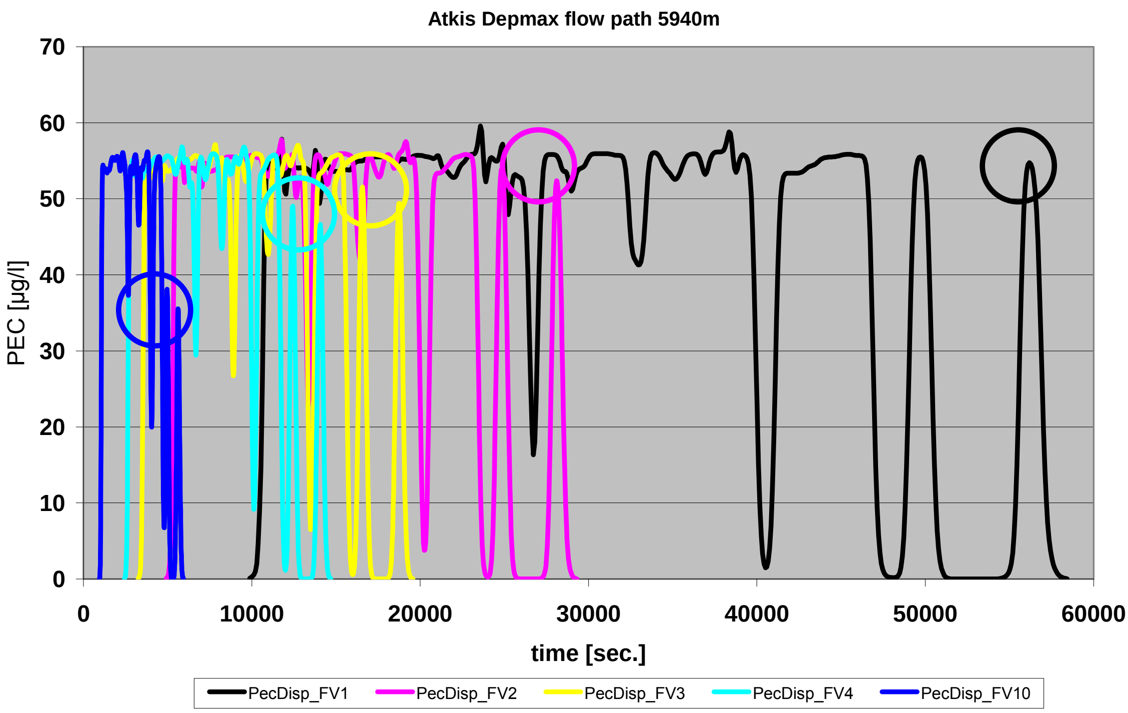

Figure 8-24: Influence of different flow velocities for the Haunsbach

The circle marks up the same entry from the first application field of the head water.

It can be seen that there is a relation between flow velocity and effects of dispersion, i.e. the higher the flow velocity, the higher the effects of dispersion.

Comparing a flow velocity of 0.1 m/s with 0.4 m/s we can see a reduction of 15 % of the simulated PEC (from 54 µg/l to 46 µg/l), comparing a flow velocity of 1 m/s with 0.1 m/s we can see a reduction of 34 % (from 54 mg/l to 36 mg/l).

In the following figures the effects of higher flow velocities of the calculation of potentially risk segments are shown. For the given depth and width of the water bodies no risk segments were predicted at flow velocities of 0.3 m/s and higher. For a discussion on remaining uncertainties and potential consequences for the risk assessment see section 8.12.

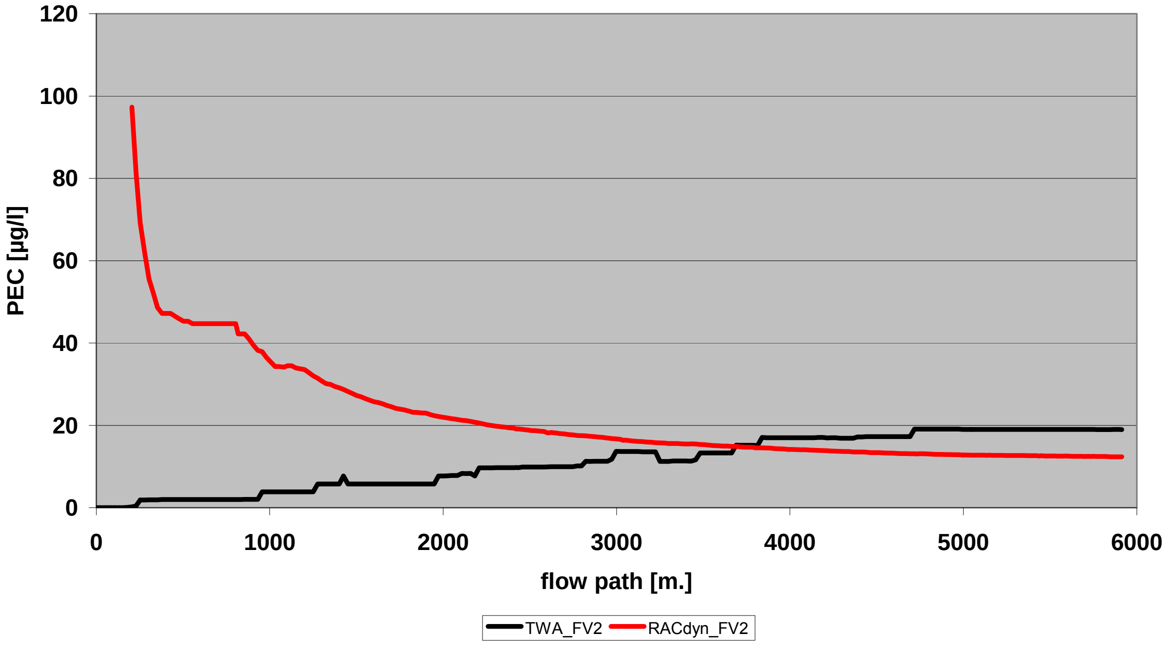

Figure 8-25: Simulation of the $RAC_{dynamic}$ using a flow velocity of 0.2 m/s for theHaunsbach

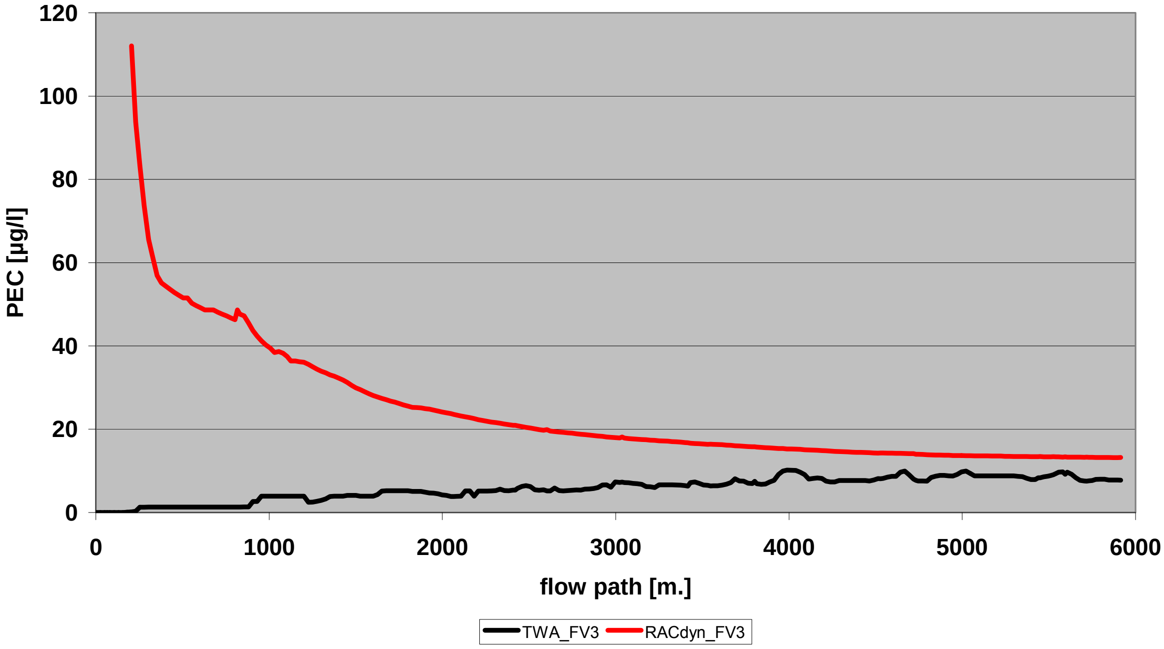

Figure 8-26: Simulation of the $RAC_{dynamic}$ using a flow velocity of 0.3 m/s for the Haunsbach

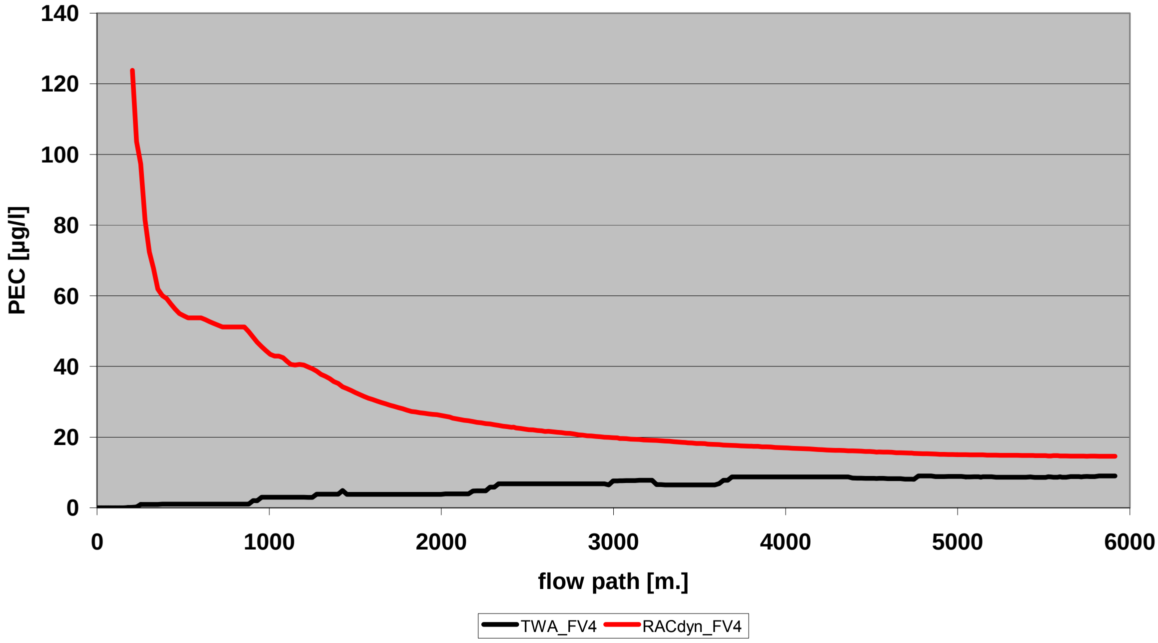

Figure 8-27: Simulation of the $RAC_{dynamic}$ using a flow velocity of 0.4 m/s for the Haunsbach

8.11.2 Effects of dilution

An additional effect of dilution is based on the higher water discharge by converging in larger water bodies.

This is demonstrated by using a simplified approach regarding only the larger water volume without effects based on vertical or horizontal dispersion.

In the investigated hop growing region Hallertau the main drainage systems are the rivers Ilm and Abens. Shortly behind the confluence of the Haunsbach in the Abens and of the Lauterbach in the Ilm, width, depth and flow velocities were measured.

Table 8-7: Hydrological parameters of Abens and Ilm

Shortly behind the confluence	Width [m]	Depth [m]	Flow velocity [m/s]
Abens	2.60	0.25-0.45	0.16-0.25
Ilm	6	0.25-0.30	0.48

Table 8-8: Effects of dilution based on larger water volume of Abens und Ilm

Water course	Depth [m]		Width [m]		TWA_max 1h		Reduction
	before	behind	before	behind	before	behind	[%]
	Confluence						
Haunsbach	0.10	0.25	0.66	2.6	19.48	1.98	90
Lauterbach	0.11	0.25	0.52	6.0	20.48	1.06	95

In the following figure the location of the relevant water courses in the hop region Hallertau are mapped. It can be seen that the main parts of the relevant water courses are small water bodies, which flow into the two main drainage systems Abens and Ilm; furthermore, Abens and Ilm do not have application fields in a buffer zone of 150 m and therefore no entry of drift related deposition occurs.

This means that drift entries into Abens and Ilm are mainly influenced by the small water bodies and dispersion and dilution effects can be considered as evident. Reduction rates of 90 % (Abens) and 95 % (Ilm) were estimated.

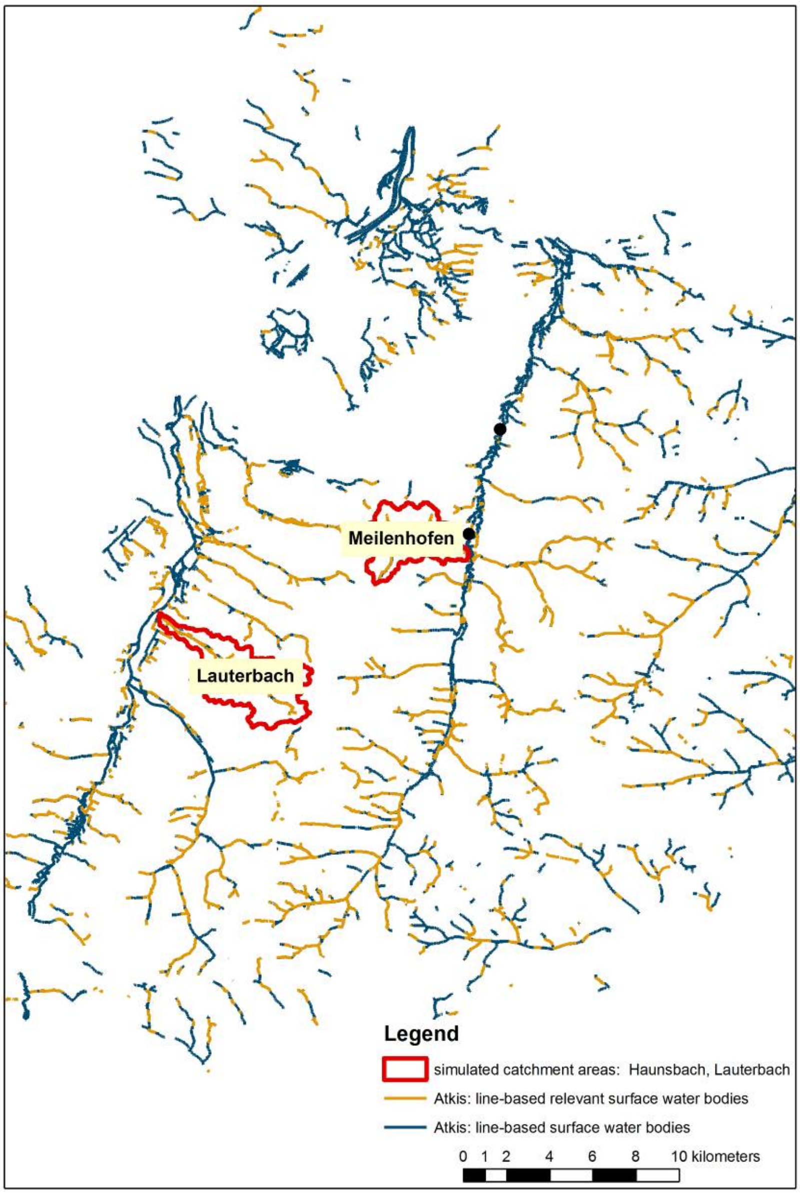

Figure 8-28: General map of the geographical situation of the relevant water courses in the hop growing region

Detailed photos of the respective main receiving water bodies are given below. The images were provided by Susanne Elbers in the context of her diploma thesis (in process). The correct location of the photos is documented in Figure 8. The photos were made directly downstream of the confluence of the investigated small surface water bodies in spring time 2009.

Figure 8-29: River Ilm nearly 5 km downstream of the confluence of Lauterbach

Figure 8-30: River Abens near downstream of the confluence of Haunsbach

8.12 Conclusions

The basic challenge was the development of methods and concepts to facilitate the dynamic modelling of drift related pesticide exposure in flowing water systems.

One necessary requirement to use the models was the existence of a consistent orographical network of water streams. This was sufficiently realised in the framework of the diploma thesis of Guerniche et al (2009).

The necessary hydrological and mathematical concepts for a dynamic modelling in flowing waters are implemented. The implementation of the algorithms in ArcGIS is realised.

The results of the simulations led to following conclusions:

- There is a differentiated exposure pattern in space and time.
- Only short-time exposure patterns were observed varying between 5 and 10 hours depending on flow path length and flow velocity.
- It was demonstrated that the choice of the percentile of the $PEC_{TWA(1h)}$ is not as sensitive as the choice of the application pattern (close range between the maximum $PEC_{TWA(1h)}$ and the Minimum $PEC_{TWA(1h)}$ of the 25 simulations).
- It was shown that the model is very sensitive to variations of the hydrological parameters. Therefore water depth and flow velocity play a major role.
- Dispersion and dilution are important especially when small streams flow into larger streams.
- Following the results of the analysis of representativeness the two streams under investigation have a typical hydrology for small streams in the Hallertau and a high index of exposure.
- Taking into account the model assumptions and uncertainties a factor of 10 could be postulated between the static and the dynamic model (regarding flowing water bodies previously treated like stagnant waters). This leads to a first assumption of 200 km of risk segments for all relevant surface water bodies in permanent crop regions in Germany when implementing the dynamic model.

Consequences for the framework of PPP authorisation:

In the past the implementation of a landscape based modeling of spray drift entries into surface water has opened new perspectives for the authorisation process of pesticides. On the basis of "real world" data proven spray drift models were taken from their test sites out to the landscape. Geo-referenced entries were calculated for real water bodies. Although this is an important step there is still missing a lot when wishing to assess the risk emerging from pesticide spray drift for waterborne organisms. In contrast to assumptions made in the first GIS-based approaches linear water bodies in central Europe seldomly are stagnant. Thus, the pesticide is transported and diluted due to the conditions of flowing water. As a consequence of this different exposure situation compared to the one in stagnat waters, magnitude, duration and location of effects are changed.

The approach at hand at least implements the movement of water and goes one step further than simply modelling of entries defining PEC_{ini} > RAC as threshold for ecological risk by taking into account a temporal component and the corresponding changes in the spatial status of PPP-concentrations (movement, dilution). This leads to the definition of a new threshold for the ecological risk in this context, namely, the "dynamic" regulatory acceptable concentration, RAC_{dyn}. In flowing waters the risk for segments exposed to spray drift can be defined as $PEC_{TWA(1h)} > RAC_{dyn}$, whereby RAC_{dyn} depends on the predicted exposure duration (here estimated as the time of the PEC above the usual RAC (ToTh) s. Chapter 6.3).

On top of this, the new approach offers new possibilities when taking into account the spatial pattern of applications in a certain area. The first approaches based on geodata assuming stagnant water also assumed simultaneous applications on all fields observed (which is conservative, but will actually never happen). The dynamic approach uses Monte-Carlo techniques to distribute the possible applications along a stream over two days. This method is far more suitable to reflect the temporal pattern of application events at the different fields along the stream.

The development of this dynamic geodata based approach marks a significant step towards a reasonable assessment of ecological risks emerging for waterborne organisms from intensive agricultural landuse. It offers a better reflection of the real circumstances and conditions determining both spray drift entries and the transport and dilution of the PPPs in the water bodies. Taking into account the real situation has crucial advantages. It makes it easier for the authorities to communicate rules and restrictions concerning the use of pesticides. In addition it might serve as a basis for the establishment of landscape-related risk management measures, e.g. restricting pesticide use where this is necessary to maintain the intended level of protection for the aquatic species and reducing product related mitigation where possible.

8.13 References

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9 Evaluation of the GeoRisk approach and proposals for implementation

Roland Kubiak, Martin Bach, , Thomas G. Preuss, Toni Ratte, Christoph Schäfers, Matthias Trapp, Udo Hommen

9.1 Introduction

Roland Kubiak

The task of this chapter 9 is to discuss the results from chapters 4 to 8 concerning their implementation into a frame concept with focus on its realisation in practice. This makes it necessary to look onto the topic from different sides:

- Regulatory and ecological impacts when the new risk evaluation and management system is brought into force and the evaluation of its socio-economic aspects (chapter 9.2.).
- Possibilities of a nation wide “hotspot”-risk management in practice (chapter 9.3.)
- Landscape related risk mitigation options (chapter 9.4)
- Implementation of risk mitigation measures as component of landscape related programmes (chapter 9.5)
- Comparison of the GeoRisk approach with deterministic and scenario-based approaches (chapter 9.6)

Since it appeared during the course of the project that two possibilities for a probabilistic geo-referenced risk evaluation were developed, it had to be decided which approach is appropriate for the registration practice.

1. The first approach is based on the use of the stagnant water body (100 x 100 x 30 cm) currently used for registration and geo-referenced analysis using the ATKIS Database (resolution of 1 : 25.000). This is described in detail in chapter 5 and the results of these German-wide calculations are given in chapter 7.
2. The second approach is based on a model concept for the geo-referenced probabilistic assessment for streaming waters. This model was developed in the course of the project and is described in chapter 4.6. This concept takes into account that the compound concentration in streaming waters depends on the timing on the applications upstream, the depth of the water body and other hydrodynamic parameters. Based on that, high concentrations may appear for quite a short time. This short time exposure of compound concentrations is considered in the risk evaluation and described in chapter 6.2.8.2. Results from geo-referenced high-resolution calculations in the hop area and at the Lake Constance region using this new dynamic exposure model are given in chapter 8.

The results of the Germany-wide calculation of hotspots in hops, grape vine, and orchards on the basis of ATKIS and the static exposure approach showed that around 2.000 km of management segments appeared (without the region Altes). A projection of the calculations made using the new dynamic exposure model came to about 200 km of management segments.

Moving from the first approach to the second means to implement more reality in the risk evaluation: firstly because of considering streaming waters and second because of a more realistic time dependent effect analysis. Both together lead to only one tenth of the hotspot segments calculated with the stagnant water body model. Beside that, the dynamic exposure model is more realistic in finding potential risk management segments since it takes into account that water depths may be significantly lower than 30 cm. However, if the data are available, also for lentic ditches more realistic water depths could be used.

Looking at the practical point of view it must be pointed out that a Germany wide hotspot management for more than 2.000 km is not possible for the stakeholders of agricultural business. The reasons for this are:

- The **manpower**: The on-site organisation would need additional staff to be paid by the federal states. On the basis of their discussions about significant cost reductions for agricultural administration, this will be not possible.
- The **money**: From studies about cost-effect relations of measures to improve the structure of surface water it can be concluded that initially 0 to 10 000 € / km and annually 0 - 6000 € / km are necessary (depending on the type of the mitigation measure, see chapter 9.4 for details). From this it can be concluded that for management of the 2000 km hotspots as predicted for all permanent crops in Germany by the static model (chapter 7) initially up to 20 Mio € and yearly up to 2 Mio € per year are necessary. However, these maximum costs for the unrealistic scenario that all hotspots would be managed by temporary fences to protect against drift. Planting hedgerows or clearing away rows of cultivated plants close to the water would result in significantly lower costs and other measures might have no or only neglible costs.
- The **time**: It would need a German-wide political initiative to manage more than 2.000 km by on-site measures. In opposite to that 200 km could be managed with a yearly amount of around 600 000 € in about 5 years[50].
- The **acceptance**: As a result from the workshop and from several talks with stakeholders from administration it can be concluded that there is only a limited willing to spend resources on hotspot management.

From all that together the consortium concluded to propose the new developed dynamic exposure model being the final GeoRisk proposal. So, all other discussions following in this chapter are based on this proposal.

[50] For this estimation the maximum costs for initial management plus 4 time the maximum yearly costs accordinig to table 9-2 were used.

9.2 Evaluation of the implementation of the GeoRisk Approach – ecological, regulatory and socio-economic aspects

Roland Kubiak, Thomas Preuss, Toni Ratte, Christoph Schäfers, Udo Hommen

To evaluate ecological and regulatory consequences of the new approach, the potential hotspots have to be quantified and related to associated landscape related and cultural parameters. This information is crucial for the discussion of local mitigation measures. At the same time, the degree of uncertainty in identifying hotspots dependent on model assumptions has to be estimated and compared with the uncertainty of the current approach.

A direct comparison of the different risk assessment approaches with respect to the sensitvity of the input parameters as well as the final protectivity of the respective approach was not possible within the projekt owing to the fact that several work was conducted in parallel and thus, not always the directly comparable assumptions were made for the different analyses. However, data analysis and calculated model scenarios within the GeoRisk project figured out some essential facts related to the uncertainty of probabilistic geo-referenced approaches as well as for the current risk assessment. For exposure, these facts include a mismatch of assumed and real wide-depth ratios for waterbodies and additionally a spatial exposure in stream systems which can be much larger than expected from stagnant water bodies. For effects, several factors were identified that become important if a more realistic but still conservative risk assessment should be applied, e.g. timing and exposure interval of exposure, extrapolation from individual to population level and from lab to field. Additionally, facts which are not explicitly or not suitably addressed in current environmental risk assessment become obvious if a more realistic risk assessment should be applied, including multiple applications (see chapter 6.4), simultaneous applications to several fields (pattern of the applications in the crop areas upstream, see chapter 4.6 and 8) and effects by mixtures (not considered within this project, see reports of ongoing UBA projects on mixture effects).

In several monitoring studies in Germany the majority of edge-of-field ditches and streams exhibit a water depth of 10 ± 5 cm, resulting in width-depth ratios of 5 – 8 for ditches and 7 – 14 for streams (Table 4-18) whereas for current risk assessment a depth of 30 cm and a depth-width ratio of 3.3 is assumed. Using a non-realistic and - more important - non-conservative water depth leads to lower calculated PECs than can be expected in the environment. Additionally it is a well established fact that the water level shows high annual variability, with the lowest water level in summer and the highest water level in spring. Since applications of plant protection products are conducted mostly in summer, the lowest level should be used for ecological risk assessment.

From the dynamic calculation of PPP concentrations in streams, it becomes obvious that for a spatial analysis a static approach is not necessarily conservative related to the exposure assessment due to two important facts related to the dynamic nature of streams. First of all, exposure to one segment also affects the downstream segments in streams. If disappearance of the compound e.g. by partitioning to sediment or degradation is not taken into account, this potentially leads to more segments with PEC > RAC by downstream flow of the exposure event. Additionally, due to the dynamic nature of streams a flowing water segment can achieve several loads of plant protection products from different fields leading to an increased PEC compared to the static approach. This is illustrated in Table 8-2, in which a higher length of hotspots is indicated using the dynamic model compared to an RAC based on standard ecotoxicity test results. Only if the RAC is adapted to short-term exposure (RAC_{dyn}), the length of identified hotspots is reduced in the dynamic compared to the static approach.

Due to the structure of the project (parallel hotspot identification and development of parameterisation) and the time needed for GIS data processing, the hotspots for pesticide use in hops, wine and orchard cultures in Germany were identified based on the static ditch model, not including the stream-related considerations concerning water depth, stream velocity, dilution, downstream transport, and short-time exposure. However, the project consortium regards the dynamic stream considerations the most realistic approach and the only reliable one for georeferenced exposure calculations in the future.

Within this project, a re-calculation of hotspots based on the dynamic model for lotic water bodies relevant for permanent crops in Germany was not possible: During the course of the project it became evident that the originally required static approach would be unrealistic for the situation in the field with a majority of running edge of field waters. It became also clear that at least the theoretical concept to consider the exposure in running waters could be elaborated. Because the dynamic approach requires additional data compared to the static one (e.g. connected stream network, flow directions, flow rates) which are not providied by ATKIS also an approach to create this data was developed (chapter 8). However, the development and application of a dynamic exposure model (and the consideration of pulse exposure in the effect assessment) was included in the original project plan. Thus, development of the dynamic approach was done in parallel to the implementation and analysis of the static approach (chapters 4 and 7) and there were neither the time nor the resources available to apply the dynamic approach for all permanent crop areas in Germany. Therefore, the quantity of hotspots to be expected based on the dynamic approach was roughly be estimated by comparing the output of the calculations according to the static model with those according to the dynamic model for streams in an exemplary worst case region. The proportion of hotspots was used to extrapolate to hops areas and all permanent crops in Germany (see chapter 8.10). However, this extrapolation was based on the assumption that all water bodies are lotic. Despite that water bodies attributed as ditch in ATKIS are not necessarily lentic, this might result in an underestimation of the hotspots if the dynamic model is applied to all lotic waters. Without the full dataset on the hydrodynamic parameters (i.e. velocity, flow volume) for all relevant water bodies, the proportion of lentic ditiches can not be assessed. On the other side, the streams used for the example applications of the dynamic model were shown to be representative for hydrodynamic properties and realistic worst case for streams in this region.

The area used for the exemplary assessment was the Hallertau, an intensively cultured hops area, where hotspot analyses according to the static model were finished early. The identified hotspots were eologically assessed during a field excursion from 9 to 11 March 2009. The specific reason of this excursion was (1) to get basic information about the evaluation of the significance of managing arrangements in the Hallertau region and (2) to find out if there are typical ditches with typical characteristic for references (no impact) and hotspots (high impact). Subsequently, five selected small running waters were investigated i.e. characteristics of the water body were measured and biological samples were taken at 14 sites.

The results obtained from these analyses together with monitoring data sets from the orchard region ‘Altes Land’ and streams in the region Hannover / Braunschweig were used to develop the trait-based approach by an analyses which traits are found at sites likely not or only slightly exposed to pesticides (reference sites) compared sites close to crop areas and thus likely exposed to pesticide entries. Finally this analysis led to the selection of three realistic worst case (macroinvertebrate) species for further analysis (see chapter 6.2). By selecting representing trait combinations found in reference sites but on lower abundance (or not at all) in sites assumed to be exposed to pesticides entries, the evaluation was based on the “realistic best case” for edge of field water bodies as defined by Wogram (in Brock et al., 2009).

Thus, the scientific basis for trait based hotspot criteria was developed but could be implemented within the project neither for the German wide static approach (see chapter 7) nor the example calculations using the dynamic approach (see chapter 8).

For two streams exhibiting the highest density of hotspots, the exposure calculation was calculated including the dynamic model (chapter 8). The extrapolation to the total German hops, vine and orchard areas depends on the applicability of the dynamic model. Whereas for streams the extrapolation is conservative due to the worst case character of the exemplary area, exposure of static ditches should be estimated by the static model, however using realistic depths. The model scenario used in the actual legislation is based on a ditch of 30 cm depth and used generically for all streams in Germany. This approach is usually considered to be sufficiently conservative as the depth estimation being approximately three times higher than depth of real edge of field waters is levelled out by neglecting dilution and transport in flowing waters which in reality reduce magnitude and especially duration of exposure. When enhancing reality of the approach by differentiation of hydrodynamics, of course more realistic depths should also be applied to static ditches. Thus, the GeoRisk approach takes these facts explicitly into account for different exposure situations and thereby decrease the uncertainty in the environmental risk assessment compared to the current approach.

Because ATKIS differentiates streams and ditches due to morphological rather than landscape profile or hydrodynamic criteria, the dynamic stream model (chapter 8) should also be applied to ATKIS ditches to identify stream velocity, discharge and flow direction.

As edge-of-field water bodies in Germany are mostly streams and ditches, whereas the risk assessment is still focusing mostly on standing water, it is recommended that also the current risk assessment should focus on ditches and streams. The application of the proposed dynamic approach will likely result in considerably reduced hotspots compared to the static approach when regarding lotic waters (see chapter 8.10). However, due to often considerably lower water volumes resulting from the use of more realistic lentic ditch depths, PEC_{ini} values will be higher and thus, probably more hotspots will be identified for lentic ditches.

Thus, including more reality reduces regulatory uncertainty due to the following aspects:

- Streams are shown to be overprotected by the static approach. By using the dynamic stream approach (for streams as well as for lotic ditches) it has to be assured that the probabilistic upstream pesticide use includes realistic worst case scenarios. When assessing risks of acute effects, the approach presented in chapter 8 is regarded safe, as
    - the reference active substance Carbaryl represents the worst case with respect to the time to effect manifestation and the difference between 1 h short term drift effects and 96 h standard toxicity data
    - Carbaryl was not included in Annex 1 due to the severity of acute effects
    - for lotic waters, 1 h of exposure is sufficiently conservative for realistic situations

  - o at shorter exposure durations toxicity seems to occur at much higher concentrations

- Regarding chronic effects the approach is also considered to be conservative because in accordance with the hotspot criteria (chapter 8.2) effects were always assumed to be lethal – x % inhibition of reproduction, growth or development are considered as x % reduction of abundance or biomass. Thus, if the RAC is based for example on the chronic *Daphnia* test (NOEC for inhibition of reproduction divided by a factor of 10) this can in a first step compared to the $PEC_{TWA(1h)}$. For refinement, a PEC_{TWA} for another time span can be used as suggested in Brock et al. (Elink workshop, 2009, for details on consideration of exposure duration see chapter 6.3).
- Lentic ditches are shown to be probably underprotected by the static approach. Here, including more realism (i.e. by separating lentic from lotic ditches and by the use of more realistic water depth values) results in a higher protection level. For lentic ditches (and other lentic water bodies, e.g. ponds) uncertainty can be reduced by higher tier ecotoxicological test methods, e.g. mesocosm studies which are specifically representative of lenthic ditches with respect to water quality (pH, oxygen levels), pesticide exposure patterns, represented species, and modes of recovery.
- The shoreline of ponds and areal water bodies like big streams is handled like in the current static approach (considering this part a lentic ditch of 1 m width and 30 cm depth, see chapter 4.4). Thus, the protection level is comparable to the existing system. This was regarded necessary, because in riparian zones of areal waters lenthic, shallow situations might occur. As these zones are especially species-rich and e.g. mating and breeding areas of aquatic vertebrates (fish, amphibians, birds), we decided to keep the existing assessment. Due to the probable dilution by contact to the deep of flowing main water body, this approach can be regarded as conservative.

9.2.1 Suggestion for determining protection level with special attention for spatial heterogenity

In general a sufficiently high protection level should be reached in all waterbodies (excluding canalized ditches from the analysis). If possible, the community recovery principle should be applied. In practice the recovery potential can only be predicted with high uncertainty for streams and isolated ponds. Therefore, if the recovery potential cannot predicted sufficiently safe, the ecological threshold principle has to be applied. However, as lentic ditches are the most critical waters with respect to exposure and driving the majority of hotspots, recovery can be included for these biotopes.

According to the protection goal given in the German Pflanzenschutzgesetz and the EC Directive 91/414/EEC, longer-lasting effects on populations and communities have to be avoided (UBA 2007); i.e. during an authorisation period of 10 years unjustifiable effects on populations due to the application of a product according to the intended use must not occur.

The assessment criterion for a certain water body section to meet or not to meet the protection goal is a decision about whether this section is a hotspot or not. Therefore, carefully and scientific-based setting of the hotspot criteria is crucial. These criteria include both the exposure estimation and the ecologically relevant information related to the protection goal (e.g. the trait approach). A similar careful examination is necessary for the selection and consequences of management measures to convert a hotspot into a section with acceptable risk. In addition to the non-target species in the water bodies, in case vegetation shall be introduced as management measure to reduce pesticide deposition on the water body, protection goals for the non-target species inhabiting this vegetation have to be discussed and considered.

Last but not least the technical performance of the applied hardware and software should be trustworthy. It appears that ATKIS yet has shortcomings since a number of water sections are not connected as they are in reality. The following discussion does not regard these problems and is based on a well performing GIS.

Validity of the hotspot definition

The hotspot definition includes a spatial aspect which has to be related to the activity range of aquatic populations, the estimation of a realistic exposure concentration and duration resulting in a PEC for each 25 m segment and a RAC. The PEC then is compared to the RAC. The scientific validity of the underlying assumptions and definitions is discussed in the following.

Spatial definition

The consortium used the hotspot criteria given in UBA 2007. A hot spot is a water body section of 1000 m in which the criterion “PEC > RAC” is true for more the 10 % of the 25 m segments or the criterion “PEC > 10*RAC” applies for one segment. The definition of the spatial extent of a hotspot is crucial for the pesticide effects on and recovery of populations, since the impact of pesticides on running water biota depends not only on the duration and spatial distribution of a PEC > RAC event but also on the extent of this exceedance. According to a literature survey conducted in the current project the hypothesis is supported that 1000 m can be seen as an appropriate measure for the activity range of most of the running-water species. Species of higher mobility with a wider range on one hand would require the definition of larger water sections but on the other hand it can be shown that the definition of larger hotspot sections would lead to fewer hotspots. Therefore, it appears that the spatial hotspot definition is rather conservative and in line with the protection goal.

If the suggested approach should allow simplification and lowering of mitigation measures while ensuring the desired level of protection of the edge of field water bodies the hotspot identification and management must be conducted for all relevant water bodies. It is not sufficient e.g. to restrict the hotspot management to “Sondergebiete” (specifically defined areas in Germany where now reduced mititgation measures are allowed, e.g. the orchard region Altes Land). The general protection goal can only be achieved if all hotspots in Germany resulting from generally reduced mitigation measures are identified and managed.

PEC estimation

The PEC estimation includes the estimation of deposition rates and modelling the receiving water body. The respective model is decisive to calculate the dilution, adsorption and transport of pesticides.

Among the most important factors determining the spray drift distribution are wind speed, wind direction, drift reducing measures (nozzles), humidity, temperature, etc.). The inclusion of these factors into the calculations is amply discussed and assessed in a transparent way in Chapter 4.2. The decision was to provide a local spray-deposition distribution (resulting from Monte-Carlo simulations for eight wind directions and considering the variability of deposition rates) from which the 90th centile is used for the PEC estimations.

For multiple applications the use of individual drift percentiles for the single application events that represent the overall 90th worst case without considering degradation/transportation between two applications is recommended (see chapters 4.2.4 and 6.4). For lentic water bodies it is suggested to use so calculated PEC_{ini} for a single application event and consider the effect of multiple application by hotspot critera using tolerable effect levels depending on the number of applications (see chapter 6.4). For the dynamic approach, the application events are simulated anyway and thus, for multiple applications the whole application period can be simulated. For effect assessment it is suggested to use the maximum PEC_{twa} to calculate the expected effects and to apply tolerable effect levels estimated for multiple applications (chapter 6.4). Dissipation of the pesticide between the application events can be considered by higher tier approaches.

The PEC estimation also considers deposition reduction by shielding vegetation at the water side and emerged vegetation of the water body itself.

Whereas the estimation of the deposition rates and the use of the 90th percentile appear reasonable and conservative, the estimation of deposition reduction is yet based on very simple assumptions.

With respect to the receiving water body, as a starting point the static water model (1 m width; 30 cm depth) was applied and PEC_{ini} calculations were performed using ATKIS data. The results of these Germany-wide calculation of RMS in hops, grape vine, and orchards showed that several hundreds km of hotspot areas are to be expected for orchards and grape vine and more than hundred km for hops.

This poses two questions: (1) Are the assumptions in the static water model realistic and reasonable, i.e. based on scientific knowledge? (2) What would be the consequences if the first questions would reveal a YES? Indeed, it appears that in these calculations quite unrealistic assumptions about the fate of a compound are made and information of the real water depth as well as processes of dilution, transformation, adsorption, volatilisation are not considered which do affect the concentration of a compound in running waters. In view of these shortcomings, neither acceptance from the scientific community nor from the agricultural advisors and farmers can be expected from this concept.

Therefore, during the present project a more detailed concept was developed, distinguishing between standing and moving surface waters and calculating a dynamic flow model of the flowing surface waters. The model allows the calculation of PEC_{ini} based on the distances between fields and surface waters, the drift tables of Rautmann and Ganzlmeier and the depth of the water body. Beside that the model allows calculations of PEC_{TWA} (PEC_{dyn}) in surface waters based on the application scenarios and the flow velocity.

For the time being the dynamic water body model is preferable, since it represents a scientific sound and transparent approach and covers the state-of-the-art of what is known about deposition and partition of compounds in dynamic water bodies. Moreover, it turns out that on this more realistic basis significantly less RMS can be expected for orchards, grape vine and hops.

Nonetheless, since some rough assumptions had to be made (e.g. concerning the interception due to the shielding vegetation) the PEC calculations according to the new model should be confronted with measured data from the field, i.e. a future monitoring project/programme is strongly recommended.

Further uncertainty is caused by the ignorance of habitat types. Since these are essential for predicting the presence/absence of species and thus the critical traits, as well as for predicting the recolonisation potential. The trait analyses conducted in the preset aspect were based on relatively few case studies and the most sensitive traits were related to single species which is seen as a worst case approach. In reality there are various habitats with more or less characteristic species assemblages. A targeted investigation of these habitat-type related species assemblages of small rivers with respect to realised trait combinations is yet missing and could not be performed in the present project, but should be performed before the implementation of the trait-based approach. It could turn out that the most sensitive traits and species identified by the current project are not relevant for other habitat types and that there are species combinations showing trait combinations which are less sensitive (or more sensitive). Further, the potential coherence between application periods and the habitat-specific occurrence of sensitive life stages is crucial.

Also a consideration of new habitats arising from possible mitigation measures due to the management of hotspots is very important. These measures have to be carefully selected, since the newly-settling species (e.g. birds, butterflies, coleopterans) could themselves fall among those species which have to be protected. However, if clearly created on agricultural areas in order to reduce drift entries into of-cropp areas including water bodies these hedges should not be considered as to be protected as off-crop area.

The mentioned uncertainties and gaps in knowledge do not allow implementing the GeoRisk approach immediately. But from a scientific point of view there is hardly another way to come to a science-based and realistic risk assessment in water bodies. The GeoRisk approach provides a framework of almost all processes and facts which have to be combined. Also the yet missing information can easily be identified. Therefore, the consortium highly recommends inducing further research to reduce the uncertainties and to implement a pilot project in which the model prognoses of a refined GeoRisk approach can be confronted with field data.

9.2.2 Coherence with other regulatory approaches

When evaluating hotspot management measures, within the pesticide regulation, there may be contradictions between protection goals concerning the aquatic communities, the terrestrial non-target plants and arthropods and the birds and mammals. Hedges implemented to minimize spraydrift exposure will enhance the structural landscape diversity and thus improve the status of the landscape with respect to nature conservation issues. At the same time, they create new biotopes that can be matter of protection themselves and enhance the probability of birds and mammals to stay near or in the field. Thus, it must be sure, that hedges planted for drift mitigation (especially those planted on land previously used for crop) do not result in new obligations for risk mitigation, e. g. spray distances, to protect the hedges. The consequence would be a loss of agriculturally used area. To enhance acceptability of hotspot management and to benefit from the structural improvement, the status of the hedges should be handled as it is for in-field structures.

Furthermore the GeoRisk approach is coherent with the following regulatory approaches:

- An interference with **WFD** monitoring is not expected due to the fact that the WFD monitoring does not include such small water bodies that are in focus of the GeoRisk assessment. As long as the substance concentration resulting from entries above RAC in small water

bodies does not cause an exceeding of EQS in the monitored WFD-relevant water bodies, the outcome of a geo-referenced PRA does not conflict to the WFD.
However, pesticide concentrations in edge-of-field streams calculated by means of the dynamic model, can be used for the estimation of pesticide concentration in larger stream sections for the WFD (Water Framework Directive, EC 2000). The water bodies used in the water quality evaluation legislation generally are of higher orders and sizes than the edge-of-field water bodies focused on by GeoRisk. It is assured by WP1 and WP2 that the pesticide freight output in higher order streams can be calculated and used as input in GREAT-ER (www.great-er.org) or DRIPS (Röpke et al. 2004) calculations. This will be a substantial improvement of the assessment of pesticide exposure from diffuse sources and will enhance the explanation of observed water contaminations.

- The geo-referenced exposure as developed in the current project should be appropriate for a link to geo-referenced habitat maps (**FFH directive,** EC 1992). In future, it will be possible to overlap exposure assessment of chemicals based on point and diffuse sources with species distribution maps used in nature conservation and apply population and community effect models for different stressors. This vision of a harmonization of different regulations (substance-, species- and landscape-specific) in focusing on the site-specific community of populations should be accounted for.

With respect to proposed management measures it should be ensured that there is no reduction of quality in the sense of the water framework directive (WFD), e.g. by filling ditches or leading streams partly underground. The aspired reduction of maximal distances to water bodies of the application of pesticides in crops has to be communicated as implementation of permanent buffer stripes guaranteeing a minimal distance of crops to waterbodies.

9.2.3 General conclusions

- The GeoRisk approach makes an important step to a more realistic risk assessment and contributes to the clarification of uncertainties. While the current approach considers uncertainties by the general application of worst case assumptions leading to a likely high level of protection but also the risk of overprotection in many cases, the geodata based probabilistic approach is more complex but allows (but also needs) more explicit decisions on handling the uncertainties. However, quantitative analysis of the uncertainties of the final risk estimation (i.e. the presence of a hotspot) could not be conducted within the project.
- The GeoRisk approach will lead to a a risk management, which is much more transparent and more easily to communicate than today because it is based on local assessments. The higher complexity of the assessment procedure is thereby included in the exposure and effect models and do not have to be addressed explicitly in the regulatory everyday life. Therefore it will be easier in future to include new scientific knowledge in the risk assessment. Risk assessment will become verifiable, which is not the case nowadays.
- The differentiation of exposure situations (and in the future potentially also species abundances) facilitates agricultural work and enhances environmental protection in appropriate areas.
- Including realistic landscape elements enables the harmonisation with other regulatory approaches (WFD, FFH directives).

9.3 Proposal for the implementation of hotspot management

Roland Kubiak & Udo Hommen

The decision of the consortium to follow the dynamic exposure model approach requires the following tasks to implement a hotspot management on the basis of landscape-related measures:

1. Measuring water body depths and speeds in the orchard, vine and hop areas.
2. Calculation of the dynamic stream water model for all orchard, vine, and hop regions in Germany, followed by the generic calculation of PEC_{ini} for the streaming waters.

This would need a new project and could be done during 1 ½ to 2 years with about 250 000 €.

Stakeholders to be involved in the whole process of hotspot management should be representatives from:

- Registration: BVL, UBA
- Administration: agricultural extension service of the federal states, local advisors
- Industry: IVA
- Associations of farmers: orchard farmers, vine-growers, hop-growers
- Science: research institutes

A hotspot management should cover the following requirements:

- All stakeholders should exactly know where the single hotspots are.
- The hotspot management must of course be carried out locally but on the other hand it must be organised and supervised on a national level to make the ongoing actions and efforts transparent to all stakeholders.
- A co-operation of all stakeholders must be organised to enable the financing of the hotspot management.

9.3.1 Administrative structure of the hotspot management

Taking into account the requirements above, the most viable solution for the implementation of the hotspot management seems to be the following proposal simplified in Figure 9-1.

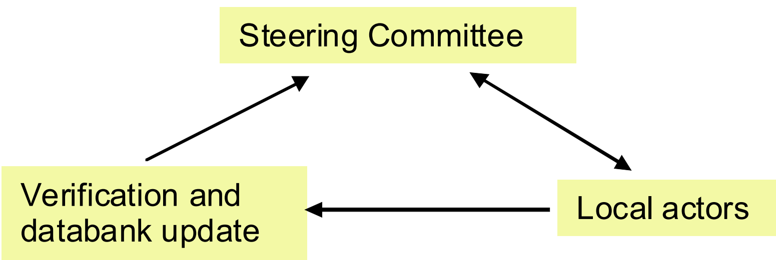

Figure 9-1: Proposal for implementation of hotspot management

The **steering committee** (SC) should be responsible for the overall organisation and the yearly budget for the hotspot management. The head of the SC should be a representative of the BVL. Members of the SC should be heads of state agricultural advisory boards, UBA, JKI, IVA, and from science.

Local actor groups will consist of agricultural advisors, members of local governments, local administration, farmers, and scientific advisors. The local actor groups should find agreements with the SC about the way to handle the hotspots and do the work.

Verification of hotspots in the field **and databank update** should be done by a scientific institute with expertise in GIS work. This institute would be responsible for hotspot identification, control of hotspot management and updates of the databank with the Geodata used for the risk assessement after a hotspot is successfully managed.

The **work flow for managing a hotspot** should include:

1. Scientific institute gives the information about a special hotspot to the SC.
2. The scientific institute makes a proposal for the risk management to the SC that decides on the proposal.
3. The equivalent head of the state agricultural advisory forms a local acting group with the help of the local agricultural advisor.
4. The local group accepts or modifies the proposal from the SC. A common agreement is made between SC and the local group.
5. The SC gives the budget for the management measures and the work is done by the local actor group.
6. After a successful end of the work, the local actor groups report the results to the SC and the scientific institute responsible for verification and databank update.
7. The institutes control the success, confirm the effort and update the GIS databank.
8. This is reported to the SC and the SC closes the file.

9.3.2 Budget for the hotspot management

Based on the assumption made in chapter 9.5 it needs about yearly 600 000 € in to manage 200 km of hotspots during 5 years. Financing in context with cross compliance is also stressed in chapter 9.5. This could be a common financing with contributions from the federal state and the plant protection industry. Contributions from the EU via cross compliance programs would have to be checked.

9.3.3 Consequences of the hotspot management for product authorization

For the product authorization described in the following it is assumed that the hotspots have been identified by a generic nation wide analysis and successfully managed to reduce pesticide entries in a way that no unacceptable effects are expected for a currently authorized worst case product.

In general two options are available for the product specific risk assessment to decide on authorisation and necessary risk mitigation:

A. Comparing a specific percentile of a PEC distribution over all (crop-relevant) water body segments with the RAC.

B. Check whether the intended use of the product would cause new hotspots in the landscape.

Option A was originally suggested during the UBA – IVA – BVL workshop (Klein et al. 2006) and was further developed in the framework document (UBA/BVL/BBA/IVA 2006), the former UBA-Project (Schulz et al. 2007) and also in a suggestion of the IVA during the GeoRisk workshop (Dechet 2009).

The general approach of option A is summarised in the following figure, details are given Schulz et al. (2009):

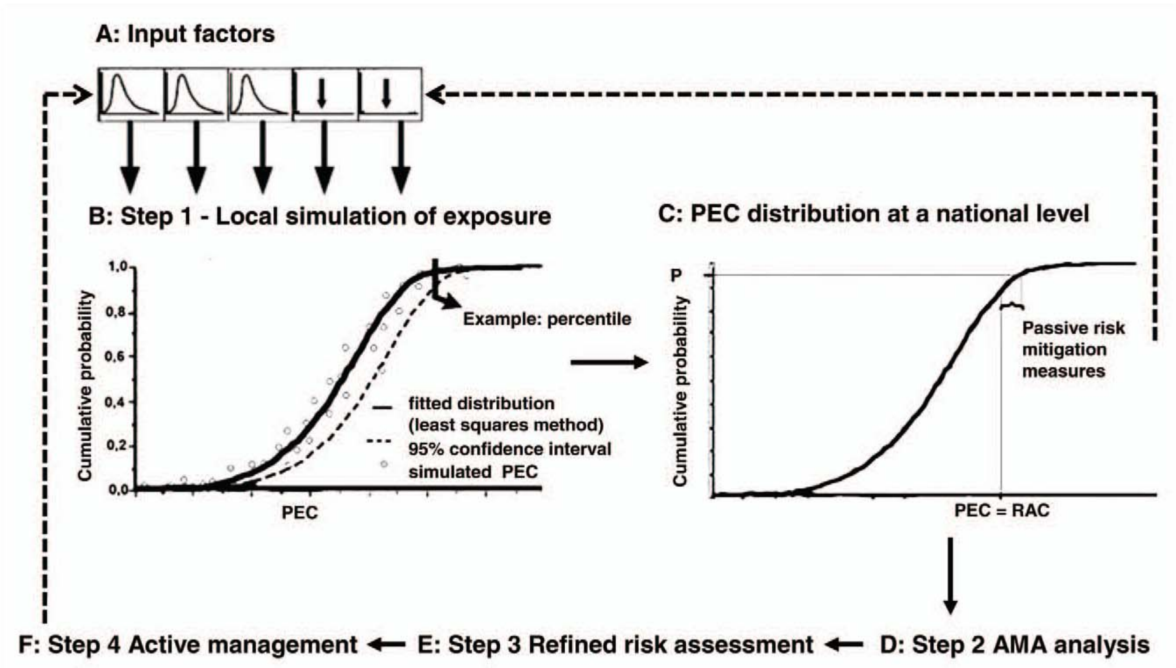

Figure 5-1: Scheme illustrating how and at which steps the setting of percentiles of exposure distributions occurs. The diagram also indicates that the number of active management area (AMA) management measures determined through a feedback step the setting of percentiles and confidence limits (from Schulz et al. 2009).

From the PEC distribution created for each water body segment (step 1 in Figure 5-1) the e.g. 90th centile is taken and combined to a new distribution, the PEC distribution at the national level. From this distribution again a specific centile is taken and compared to the RAC.

Some advantages and disadvantages of option A are presented in the following:

- Different risk mitigation options result in different landscape level PEC distributions and thus, similar to the drift tables used now, tables with landscape level PEC-values for the different combinations of crop, distance measure and centile can be created.
- These tabled PEC-values ('geoEckwerte_Aquatik', Dechet 2009) can then be used in the same way as the PEC values calculated in the current scenario based approach from the drift tables, e.g. by calculating a TER and comparing it to the specific trigger.
- The advantage of this approach is its simplicity for the user – the PEC values used for the risk assessment can easily be calculated from the tables and the application rate.

- However the disadvantage of this approach is that the spatial information is lost in the risk assessment and no link between the hotspots managed and the segments with PEC values above the RAC exists.
- In addition it seems not clear how the effects of a hotspot management should be considered in this approach. Successful hotspot management would affect the landscape level PEC distribution, i.e. the distribution will be narrower because exposure of the segments with the highest PEC values is reduced. Thus, the centile used for the comparison with the RAC will also be lower if the hotspot management is considered in the risk assessment. This would allow the use of more toxic substances (lower RAC) but would also induce the need for a new hotspot management to protect these new segments with PECs above the RAC. Keeping the PEC-distribution based on the situation before the hotspot management seems not to be an option because the database must be regularly updated and refined.
- The approach is also difficult to use with the dynamic exposure model where exposure is characterised by magnitude (PEC_{TWA}) and duration (ToT). Consideration of the very short exposure in lotic waters is not possible because the relevant RAC_{dyn} is not a fixed value as the RAC but a spatially explicit parameter depending on the segment specific ToT.

Option B avoids the use of a landscape level PEC distribution by using the identification of hotspots due to the use of the product as the decision criterion. This would ensure that there are no product related unacceptable effects on local populations and an update of the geodata base (due to changes in landscape or availability of better geodata) does not affect the criterion.

However, testing the occurrence of new hotspots caused by the intended use of a product requires the calculation of the PEC values per segment, the transfer into effects and the application of hotspot criteria. Thus, the stepwise and spatially explicit risk approach of option B needs a more complicated tool as option A which is based on a simple table.

The tool must include a database of the expected generic PEC values for different product related risk mitigation measures and the option to calculate the product specific PECs from the application rate as well as the effects per segment based on the RAC, a slope of the dose-response curves and information to consider pulse exposure. In addition, the tool must allow for the hotspot identification (with the proposed criteria developed here but also with refined product, respectively taxon specific criteria).

In consideration of the regulatory needs of a geo-referenced risk assessment and the potential link to other legislative contexts in terms of local action plans, the GeoRisk consortium proposes option B, risk assessment based on the probability of new hotspots due to the use of a product, for product specific risk assessment of spray drift in vine, fruit and hop cultures in future.

A proposal for a new system of risk mitigation measures was out of scope of the project. However, the generic assessment and the hotspot identification were conducted under the assumed objective to replace the current 20 m distance measure by a maximum of 10 m. If one or two additional smaller distance measures (e.g. 3 and 5 m) should be introduced has to be decided separately as well as the classes of drift reducing techniques.

One proposal would be 3, 5 and 10 m distance measures which could be combined with 75 and 90 % risk mitigation by the specific application techniques.

9.3.4 Registration during the time of hotspot management

The registration based on the new calculation model can be brought into force and be applied for all areas except the hotspots. For the hotspot areas the way of calculation currently in use and the application distances (buffer zones up to 20 m) should be further valid until the hotspot management is successfully carried out. After five years all hotspots should be managed and the new system is into force without exceptions.

9.3.5 Trouble shooting

Following the discussions of the workshop, the hotspot management itself should be out of the official registration process. Nevertheless the registration process can refer to the hotspot management measures and allow a reduced application distance (up to 10 m) where no hotspot exists. This is equivalent to the “Biotopindex”, already into force in Germany.

9.3.6 Other options of hotspot management

The consortium sees no other options for combining the tasks necessary for a correct hotspot management. The advantage of the nation wide hotspot management proposed is that a Steering Committee with members from registration authorities overviews the whole process and that the local management itself is carried out by the farmers themselves, the agricultural advisors and the local administration.

9.4 Local landscape related risk mitigation options

Martin Bach

Overview on mitigation options

Fundamentally, a number of measures are available for a substance-independent management of water stretches with a high risk of spray drift entry; they are listed below (Table 9-1). In this section, we address only landscape-related measures; other measures such as drift-minimizing sprayer equipment technology are not considered.

- Clearing away rows of cultivated plants close to water
- Planting hedgerows
- Establishing natural river bank strips
- Erecting (temporary) fences to protect against drift
- Temporal shifting the moving of embankment
- Neglecting the moving of embankment
- Improving the re-colonisation of river segments (as a potential component of a higher tier pesticide registration procedure), for example via establishment of regeneration zones in and around the water body.

These potentially feasible measures have to be evaluated from various viewpoints:

- Effectiveness – percentage of abatement of spray drift deposition (reduction sufficient?).
- Practicability – technical and organizational effort needed for implementation?
- Costs – amount, efficiency, one-time or yearly recurring costs, what is the funding agency?
- Acceptance – from those who are affected (mostly meaning the farmers)?
- Evaluability – verification of the implementation and effectiveness of a measure (possibly as a justiciable component of a pesticide authorization)?
- Time-frame – how long takes the implementation of the measure and what is the duration of use of an (natural or technical) installation?
- Responsibility – which private agency or public administrative body is responsible for organization, implementation, and maintenance (on the national, regional, and/or local level)?

The landscape-related measures are rated in Table 9-2 according to the aforementioned aspects. The evaluation is based on the discussions of the UBA workshop in November 2009 as well as on the previous workshop in 2007 (Schutz *et al.*, 2008).

Table 9-1: Compilation and evaluation of measures to reduce the spray drift deposition into surface waters along river segments with a high risk of spray drift entry, the so-called management segments (cf. Schulz et al., 2008; UBA-Workshop Nov. 2009; modified)

<table>
<tr><th>Measure</th><th>Effectiveness</th><th>Practicability</th><th>Costs</th><th>Acceptance</th><th>Evaluability</th><th>Time frame</th><th>Responsibility</th></tr>
<tr><td rowspan="2">1. Clearing away rows of cultivated plants close to water</td><td>Very high</td><td>Easy</td><td>Costly</td><td>Very little</td><td>easy</td><td>Short-term</td><td>Farmers, LCA[a]</td></tr>
<tr><td colspan="7">Costs mainly caused by loss of productive land (loss of revenue)</td></tr>
<tr><td rowspan="2">2. Planting hedgerows</td><td>High</td><td>Easy</td><td>costly</td><td>? [c]</td><td>easy</td><td>Short-/ medium-term</td><td>Farmers, LCA[a]</td></tr>
<tr><td colspan="7">Effectiveness depends on features of landscape elements: height and density of hedges
Time frame of implementation: short term; achievement of effectiveness: medium-term</td></tr>
<tr><td rowspan="2">3. Establishing natural river bank strips</td><td>High</td><td>Medium</td><td>Costly</td><td>? [c]</td><td>Easy</td><td>Short-/ medium-term</td><td>Farmers, LCA[a]</td></tr>
<tr><td colspan="7">Time frame of implementation: short term; achievement of effectiveness: medium-term</td></tr>
<tr><td rowspan="2">4. Erecting (temporary) fences against spray drift</td><td>High</td><td>Medium</td><td>Medium</td><td>? [c]</td><td>Easy</td><td>Short term</td><td>Farmers</td></tr>
<tr><td colspan="7"></td></tr>
<tr><td rowspan="2">5. Shifting the mowing of embankment</td><td>Medium</td><td>Easy</td><td>None</td><td>? [c]</td><td>Easy</td><td>Short term</td><td>WMA[b]</td></tr>
<tr><td colspan="7">Effectiveness depends on features of landscape elements: height and density of natural river bank vegetation</td></tr>
<tr><td rowspan="2">6. Neglecting the mowing of embankment</td><td>Medium</td><td>Easy</td><td>None</td><td>? [c]</td><td>Easy</td><td>Short-/ medium-term</td><td>WMA[b]</td></tr>
<tr><td colspan="7">Reduction of costs (saving)
Effectiveness depends on features of landscape elements: height and density of natural river bank vegetation
Time frame of implementation: short term; achievement of effectiveness: medium-term</td></tr>
<tr><td rowspan="2">7. Improving the re-colonisation[a]</td><td>?</td><td>difficult</td><td>Costly</td><td>Good</td><td>Easy/ difficult [d]</td><td>Long term</td><td>WMA[b] LCA[a]</td></tr>
<tr><td colspan="7">Re-colonisation is used as generic term for severel individual measures, e.g. establishing habitats or regeneration zones in rivers, enhancing the geomorphological structure etc.</td></tr>
</table>

[a]) LCA: Land Consolidation Authority (*Flurbereinigungsbehörde)*

[b]) WMA: Watercourse Maintenance Association (*Gewässerunterhaltungsverband, Wasserwirtschaftsverwaltung)*

[c]) Little is known about the acceptance of these measures by the respective protagonists (i.e. permanent crops growing farmers, watercourse maintenance associations, agricultural administration etc.)

[d]) Evaluability of implementation: easy; evaluability of effectiveness: difficult

Cost and other aspects of mitigation measures

Among the other aspects valuated in Table 9-2 the expected costs of diverse measures play a central role in the discussion of the implementation of a probabilistic risk approach in Germany. As a starter on this item, for the most relevant measures to reduce spray drift deposition along management segments. Table 9-3 lists the costs per unit and totaled for Germany. The aggregation for Germany is based on the calculation of nearly 2000 km of water segments to be managed in permanent crop regions in Germany derived by GIS analysis for the static exposure model (cf. Chapter 7; footnote to Table 9-2). It has to be pointed out that the calculation of MS length leads to different results when the analysis uses the dynamic exposure model to identify the water segments to be managed. At present, the difference can be specified only for hop growing regions in Germany. According to Chapter 8.10 the static exposure model accounts for 141 km MS (from 1019 km relevant water bodies), while treating the water bodies as a dynamic system reduces the length of MS to nearly 10 % of the static value. A similar comparison for the fruit and wine growing regions currently is not possible due to the lack of the MS length of the dynamic exposure model for the respective regions.

In case of diverging future results on river length to be managed - may be as based on the dynamic exposure model, or may be as an effect of an improved GIS-analysis using high resolution data – a re-calculation of the total costs for Germany is simply done by multiplying the costs per unit with the respective river MS yardage.

When discussing the total costs for Germany of various measures the reader has to be aware that the extrapolation presented in Table 9-2 is based on the assumption of 2000 km river segments to be managed. This figure is derived from GIS analysis with the criterion of the static exposure. A calculation of MS length based on the dynamic exposure assessment might lead to a significant smaller length.

Additional aspects, which have to be taken into account when establishing these measures are mentioned below. Furthermore, some 'institutional options' to embed the implementation and to fund the costs are presented in Chapter 9.5.

Table 9-2: Estimation of costs of measures to reduce the spray drift deposition into surface waters along river segments (overview, for more details see text)

Measure	**Unit**	**Cost per unit**		**Total costs in Germany**[a]	
		Initially (one-time)	**Recurrent** (annually)	**Initially** (one-time)	**Recurrent** (annually)
1. Clearing away rows of cultivated plants close to water	1 row, 100 m length	200 - 400 €	100 – 600 €/a	4 – 8 Mio €	2 – 12 Mio €/a
2. Planting hedgerows	100 m	200 – 400 €	10 – 30 €/a[b]	4 – 8 Mio €	0.2 – 0.6 Mio €/a[b]
3. Establishing natural river banks	100 m	none	none	none	none
4. Erecting (temporary) fences to protect against drift	100 m	500 – 1000 €	50 – 100 €	10 – 20 Mio €	1 – 2 Mio €
5. Shifting the mowing of embankment	100 m	none	negligible	none	negligible
6. Neglecting the mowing of embankment	100 m	none	none	none	(indirect follow-up costs?)

a) Extrapolation to Germany is based on 660 km management segments (MS) in wine growing regions, 1200 km MS in fruit growing regions (region "Altes Land" excluded, and 140 km MS in hop (rounded figures). All MS values are derived for the static approach (ref. Tables 7-1 to 7-3).

b) Maintenance of hedgerows needs to thin out the line every three to four years. The costs for one time thinning are averaged over four years.

1. *Clearing away rows of cultivated plants close to water.* The initial costs are the expenses of tree cutting and clearing of the river adjacent row of fruit, wine or hops plantation (acc. to KTBL, 2006, Table 4321). The recurrent costs cover the loss of profit contribution (gross margin) of the cleared row of formerly productive plants. The profit contribution may spread over a wide range, typical values are ca. 4000 – 6000 €/hectare for wine (BMLFUW, 2008; SWF, 2010), ca. 1000 – 4000 €/hectare in fruit tree production (BMELV, 2009; BMLFUW, 2008; in Germany: mainly apple production), and ca. 3200 – 3600 €/hectare in hop (LfL, 2010). To convert the loss of profit contribution from € per hectare to € per unit, a width of 10 m of cleared land is assumed (one to two cleared rows of fruit trees or hop, three to four cleared rows of vine).

   In comparison to the other measures, clearing is by far the most expensive alternative due to the high recurrent costs. Furthermore, a funding of this activity by agricultural schemes or other programs seems to be problematic (cf. section 9.5). Experiences with the implementation of clearing the last row(s) exist from the fruit production region "Altes Land" (northwest Hamburg). However, there the willingness of the farmers to implement this measure (clearing fruit tree rows) was massively encouraged via the alternative that the farmers would forced by legislative acts to phase out the pesticide treatments of the orchards completely.

2. The start-up costs of *planting hedgerows* are estimated acc. to KTBL, 2006 (Table 6251). The costs depend hardly from the prize for young plants: the smaller the planting material is, the lower is its prize, but on the other hand it takes more time before the plants have grown up and form a dense-close hedge. The recurrent costs are the maintenance of the hedgerow which means to thin out the line every three to four years. For the calculation in Table 9-2 the costs for one thinning out by tractor-driven machinery (40 – 120 €/100 m; cf. KTBL, 2006, Table 4121) are averaged over four years.

   By planting hedgerows (on the farmer's ground) the danger might be seen that after several years of growing, the hedgerow becomes the status of a protected landscape element which may no longer be cleared (e.g. in case that a plantation is not longer cultivated with a permanent crop and instead of this it is used for field crops whose pesticide treatment is less restricted). However, this problem also occurres in the context with other agricultural environmental programs and has been alleviated via special regulations for such landscape elements.

   As an alternative to establishing (semi-)natural hedges the plant varieties of which are typical for the vegetation at the specific site one could consider planting (and later using) tall, perennial energy crops (e.g. *Miscanthus*). However, at this stage no experience with the practicability is given, and the usage of a 5 or 10 m strip at the field edge for energy plants seems to be only a theoretical option.

3. The main difference between the measures *establishing natural river bank strips* and *hedge planting* lies in the ownership and responsibility of the measure: "Hedge planting" as to be understood here takes place on private land (mainly in the ownership of the farmer itself), while the river bank is (in the most situations) possessed and managed by the local water authority or maintenance association.

   Furthermore, "hedge planting" means the active planting of a new hedgerow, for instance with selected wood species which grow rapidly and has a high potential to reduce drift deposition. In contrast, "establishing natural river banks" focuses on the re-vitalization of the existing vegetation in the river bank towards a dense natural shielding barrier to the water formed by autochthonous shrubs and high-growing perennial herbaceous plant. This re-vitalization is a passive 'measure' initialized by the omission of annually repeated cutting (mowing) of the bank strip vegetation.

4. The use of *drift protection fences/nets* may serve as an option during a transition phase whenever newly established vegetation elements such as hedges or bank vegetation are not yet sufficiently developed to shield the water surface effectively. A second options may be a situation where the space between the last permanent crop row and the water surface is too small to establish a vegetation barrier. These fences one have to imagine similar to snow bank protection guards which are installed during the winter along roads at wind exposed sections. The initial cost of *drift protection fences* or nets comprise the purchase of the equipment, whereas the price of snow guard fences is used as reference for spray drift safety fences. The recurrent costs are the costs of work for the erection of the fence at the beginning of spraying season and dismantling at the end (cf. KTBL, 2006, calculation is based on costs of labour of 15 €/h)

5. A *shift of the mowing of embankment* means to wait with mowing of the vegetation on the embankment up to the late summer or early autumn when all spraying activities in the permanent culture are finished. However, the measure is only effective in case of presence of a dense high-grown vegetation on the embankment (e.g. reed, herbaceous plants).

6. *Neglecting the mowing of embankment* or at least reducing the frequency of moving causes no initial or recurrent costs. On the contrary, the omission of this measure saves money (ca. 30 – 60 € per 100 m embankment length; cf. Bauer et al., 2002) for the institution which is responsible for the watercourse maintenance. On the long-time perspective, neglecting the mowing of embankment for many years may end up in a very dense vegetation which forms a serious resistance to water flow in situations with high river discharge. Therefore, the water discharge capacity of the respective water stretch will be reduced significantly and will induce a heightened water-table in the water body. As effect of an increased water-table an enhanced risk for flooding of the adjacent land in case of floodwaters may occur. Furthermore, the seepage is lifted up, and the outflow of tile drainage systems, which drains into the water body can be constrained. Whether these changes form indeed relevant risks for the watercourse and the usability of the adjacent land, or just are theoretical fears, this can be judged only in spite of the specific situation of a water stretch.

Some stakeholder proposes as another measure to *omit the pesticide treatment of the row(s) close to the water*. However according to personal communications of contacted farmers as well as pesticide advisors this measure is rejected definitely, because these part of a field then forms a reservoir for animal and plant pest organism where from the main field area can be re-infected continuously. As result, increased treatment efforts are to be altogether feared.

9.5 Implementation of risk mitigation measures as component of landscape related programmes

Martin Bach

Landscape as a whole, agricultural used land, nature areas, and especially rivers, its protection, amelioration, and ecological improvement as well as the regulation of conflicts and impacts among these types of land use are objective of numerous policies and programmes in Germany and the EU. The most relevant programmes and activities which offer an opportunity to realize one or more of the drift reducing measures mentioned in Table 9-2 are listed and commented below (for reasons of clarification for Germans readers the German terms of the below-mentioned programmes, institutions, etc are added).

- *Watercourse Maintenance (*Gewässerunterhaltung*,* Gewässerpflegeplan)
- *River restoration (*Gewässer-Renaturierung*)*
- *Agri-Environmental Schemes (*Agrarumweltmaßnahmen, Kulturlandschaftsprogramm*)*
- *Land Consolidation (*Flurneuordnung, Flurbereinigung*)*
- *Landscape Management and Development Schemes (*Landschaftsplan, Landschafts-pflegeplan, Naturschutzfachplanung, Naturschutzprogramme, Grünordnungsplan, etc.*)*
- *Landscape Conservation Support Plan as part of the Impact Regulation (*Landschafts-pflegerischer Begleitplan im Rahmen der Eingriffs- und Ausgleichsregelung)

Due to the federal structure of Germany the responsibilities for the agriculture as well as the environment policy (water, landscape, and nature protection) belong to the Federal States. This state-wise competences cause a manifold of concepts and solutions with different definitions and limitations of responsibilities between the administrative bodies among the 16 German Federal States. In the following some general aspects and features can be touched briefly.

Background and further aspects of the programmes

The <u>maintenance of watercourses</u> is a legislative obligation stated by the Water Acts of the Federal States (*Bundesländer*). The institutional organisation of water maintenance for smaller rivers, brooks, and ditches (*Gewässer II. Ordnung und kleiner*) varies among the German Federal States: in some Federal States it is an obligation of the Water Authorities while in others the Watercourse Maintenance Associations (*Gewässerunterhaltungsverbände)* are responsible. Despite the type of organisation it has to be stated that from the beginning the watercourse maintenance was basically targeted to reduce the risk and minimize the damages of flooding (kernel term: "schadloser Hochwasserabfluss"). The most important operations to guarantee this goal are dredging, de-silting and management (cutting) of the emerse and the bank vegetation, inclusive trimming of trees.

Since its implementation, the watercourse maintenance operations have to comply with the requirements of the Water Framework Directive (WFD). The WFD calls for the reinstatement of water bodies and, as far as possible, for a reduction in interference in the river quality. There is an ongoing debate on the requirements of an environmentally acceptable and sustainable watercourse maintenance (e.g. Kollmann, 2004; Reinhardt, 2008; Wolter, 2007). From the scientific point of view, it is indisputable that in future before a maintenance action (dredging, de-silting, mowing, etc) takes place it has to be proofed very carefully for the respective water stretch whether an action is indeed necessary, or wheter it can be neglected or the timing can be shifted (Stiller & Trepel, 2010). Only in situations where it can be demonstrated convincingly that a maintenance activity is in the public interest, then it should be realized.

In the comprehensive survey obviously there is a big potential to install the measures "establishing natural river banks", "shifting or neglecting the mowing of embankment" and "improving the re-colonisation" as part of the maintenance. With an environmentally sounded and economically sparsely interpretation of the watercourse maintenance a lot of benefit for the river quality in general and the prevention of spray drift entries especially can be achieved.

Nevertheless, in the context of drift mitigation measures two major problems have to be mentioned: (1) Currently there is no legislative or administrative link between the water administration and the pesticide authorization, which means that the agricultural administration or the pesticide regulation authorities in Germany have no lever to force the water authorities or maintenance associations to change their practice. Actors and stakeholders involved in the watercourse maintenance have to be convinced of the challenges and opportunities of a justified maintenance, which might be a long-time process. (2) The local site conditions of water segments do not allow the abandonment of maintenance operations at all situations.

It should not concealed that the targets of pesticide drift entry mitigation may contradict to the point of view of some of the watercourse maintenance managers as well as the riparian land owners. The former group of people may have vital economic interest to generate income by executing maintenance operations, and furthermore some define their personal self-conception as an watercourse operator in terms of dredging, mowing, and cutting. The latter group often fears that the usability of their agricultural land will be constrained.

River Restoration/Rehabilitation Programmes aim to achieve the recovery to a fully (restoration) or partly (rehabilitation) working fluvial system. The measures themselves are manifold and depend on the site and the character of a water body. In contrast to the (more or less frequently) maintenance a river restoration means a fundamental intervention in the structure and morphology of a water body (e.g. the reintroduction of meanders in regulated rivers; restoration of linear transferability of rivers). While the installation of a wide natural river bank is a constituent element of a restoration (Friedl & Mohaupt, 2008), typically it is a benefit for drift reduction. Rehabilitation and enhancement of rivers is already well established in Germany over decades. Since the water policy in Germany focuses on the Water Framework Directive implementation, the restoration programmes become more directed towards the achievement of the holistic WFD targets, and singular aspects like drift reduction along some water segments are lower-ranked. On the other hand River Restoration Programmes are the only instrument to improve the re-colonisation potential at a watercourse (e.g. by establishing habitats or regeneration zones, enhancing the geomorphological structure, etc), which might be an option for spray drift risk mitigation for a higher tier GeoRisk approach.

The goal of Agri-Environmental Schemes is to integrate nature protection and conservation into agricultural production. Farmers and other bodies responsible for land management are incentivised to manage their land in a manner sympathetic to the environment by compensation payments. The schemes are installed by the Federal States, and a large number of regionally programme types and a wide spectrum of specific regulations are offered in each State. In contrast to all other programmes discussed here the targets of the schemes are shaped by the agricultural administration. This fact would simplify the integration of specific drift reduction measures. Nevertheless, the acceptance of farmers of Agri-Environmental Schemes and the participation still can be increased (Niens & Marggraf, 2010).

Land consolidation is a planned readjustment and rearrangement of land parcels and their ownership. It is mainly applied to form larger and more rational land holdings, but regularly implies measures improving the environmental sustainability, too. Typically, the implementation of a Consolidation Scheme takes 5 to 10 years, therefore it offers just a mid-term opportunity to install drift reduction measures. Additionally, only a small fraction (a few percent) of agricultural land is applied at the same time, and the activities focus on field crop regions. Thus in its practical application the potential of Land Consolidation is minimized drastically in terms of spray drift risk management in permanent crop regions.

Landscape Management and Development Schemes cover a wide range of individual programmes. In the most cases these schemes are organized and installed by the Federal States and/or municipalities. Regularly they are implemented as an obligation of EU Directives (Flora Fauna Habitat, NATURA 2000) or the German Federal Nature Conservation Act (*BNatSchG*), and/or as realization of the regional planning. Due to the legislative obligations these schemes are based on they are mainly directed towards the conservation or enhancement of individual floristic or faunistic species, specific habitats and biocoenosis, or biotic landscape elements, respectively. That these programmes could be used to install pesticide spray drift reduction measures along water bodies seems possible only when the overall river quality is ameliorated as result of the respective measure. Typically Landscape Management and Development Schemes are conceptualized and managed by the nature conservation administration. Presumably the nature conservation administration is not very disposed to operate programmes which enable the agriculture to intensify pesticide spraying.

Landscape Development Plans are implemented as a component of larger infrastructure building or settlement projects. As being part of the impact regulation the measures of these plans intend to compensate the loss of environmental quality of the land area which is impacted by the building activity. Thus the measures often have to be installed in the spatial context or in the direct neighbourhood to the deteriorated area which limits their availability to be used as drift reduction measure along rivers in permanent crop regions.

Funding options

- The *maintenance of smaller* watercourses mainly is financed by the municipalities, districts or the beneficiaries of the maintenance which are mostly the land owners within a river catchment (the regulations differ among the German Federal States). As long as a drift protection measure would not cause costs for this bodies the question of funding won't arise and the acceptance of measure implementation is not a problem of financial burdens.
- *River renaturation* is mostly financed by the Federal State together with the municipality where a rehabilitation measure takes place. While the activities don't use EU money, the States has a large freedom to target renaturation programmes in their own way. Thus, if the political willingness is given there is no obstacle to fund drift reduction measures as a part of these programmes.

- Among the programmes listed here the *Agri-Environmental Schemes* has by far the largest financial volume. These Schemes are 50-by-50 % co-financed by the EU and the respective Federal State. The programmes and its individual measures run under control of the EU Commission and has to fulfill the obligations of the EU directives they are based on. In general a measure must not grant a subsidy for agriculture. Each scheme has to be submitted and approved by the EU Commission. The authors of the report presented here cannot decide whether drift mitigation measures (which might be installed mainly with the intention to "simplify spraying") would be classified as admissible parts of an Agri-Environmental Scheme or not. This question has to be proofed very carefully before one starts to discuss this option in detail.
- *Land Consolidation* is funded by the Federal States and usually falls into the responsibility of the Ministry of Agriculture. As from 2004, the costs of assistance to planning and negotiation and to registration of alterations of property are co-financed by EU through the rural development programme. A serious fraction of the budget of a Land Consolidation action frequently is reserved for water rehabilitation and restoration measures. Hence, the realization of drift mitigation measures along brooks and rivers as a part of Land Consolidation is unproblematic from financial point of view.
- The *Landscape Management and Development Schemes* in most cases are financed by the Federal States and partly by the municipalities. Similar to the river renaturation programmes the objectives and measures of landscape related scheme are formulated by the State government (here: mainly the nature conservation administration) so in principle drift mitigation measures can be funded from these programmes.
- The *Landscape Development Plans* as part of the Impact Regulation have the advantage that the money for landscape related measures comes from private hand, i.e. the investor who impacts the environment and therefore has to pay a compensation charge for the deterioration of landscape functionalities.
- Since 2007 the EU provides a programme which *stimulates Agricultural Producing and Marketing Organisations* (EU Commission, 2007). One component of the programme funds the environmental improvement of production processes. Recently no experience is given with the application of the programme on landscape related measures. It would be interesting to figure out whether the implementation of drift reduction measures is captured by the programme targets.

A sponsorship of the funding of measures by the Pesticide Industry is strictly rejected by the German Association of PPP producing companies (*Industrieverband Agrar, IVA*). The companies hold the opinion that they have no economical benefit of a diminishment of the distance constraints or a simplification of the constraints, as the effect triggers only a limited number of compounds and the increase in compound market sales will be negligible.

As an alternative solution it was proposed that the farmers disburse the measures (or at least a major percentage) by themselves. Beside the small probability of such a solution it causes the problem that various measures unfairly burden farmers financially whenever those measures are implemented on their land at their expense. However, all farmers benefit from this whenever an effective substance can be introduced with few constraints.

Institutions

For the implementation of the programmes and schemes mentioned above various institutions, administrative bodies and stakeholder groups are involved.

- *Water authorities (*Wasserwirtschaftsämter)
- *Watercourse Maintenance Association (*Gewässerunterhaltungsverbände)
- *River neighbourhood (*Gewässernachbarschaften)
- *National and Federal State Agricultural authorities (*Agrarverwaltung des Bundes und der Länder*)*
- *Land Consolidation Authorities (*Flurbereinigungsbehörden, Ämter für Bodenmanagement*)*
- *Municipality administration (*Gemeindeverwaltungen*)*
- *Landscape Conservation and Management Organisations (*Landschaftsverbände, Landschaftspflegeverbände)
- *Nature Conservation Organisation and other NGO's (*Naturschutzverbände*)*
- *Agricultural associations, e.g. Farmers Association, Agricultural Producing and Marketing Organisations (*Bauernverband, Erzeugerorganisationen)

A cross-tabulation of the programmes with the institutions ("which institution is involved in what programme") is not very helpful due to the different assignment of responsibilities in the German Federal States. The kernel problem are less the interlaced responsibilities but is rather the fact that (i) the authorities involved in the pesticide registration on the national level in Germany (*UBA, BVL, BBA*), and (ii) also the Pesticide Advisory Services (*Pflanzenschutzdienste*) on the level of Federal States, both parties are regularly <u>not</u> involved as stakeholders in the planning and operation of the relevant programmes. Targeting, funding and operational details of the programmes mainly are specified by the water and/or nature conservation administration on the Federal State level (exception: agri-environmental schemes), whereby the agricultural and especially the pesticide advisory services regularly do not participate. As long as no legislative or political advices are given to the water and/or nature conservation administrations, they have no intrinsic motivation to open their programmes, schemes, activities etc. (and to share their sparse funding) for the demands of agriculture to implement drift reduction measures.

As conclusion it has to be stated that at present no institution feels responsible to establish risk mitigation measures or to start an initiative towards this target. For further risk management measures in areas with permanent crops, it appears vital to bring together all relevant stakeholders to a round table and mutually develop a concept delineating which responsible parties will be able to implement which risk mitigation measure at which location. In this context it would be helpful to find out whether the German Ministry of Food, Agriculture and Consumer Protection (*BMELV)* has considered integrating risk management measures into the legislative principles (e.g. into the pesticide authorization procedures or national action plan) as well as perhaps executing and financing these measures at the federal level.

In order to introduce and implement spatially localized measures, experiences from similar fields could be made useful. For evaluating the EU cross compliance requirements, several practical and effective solutions have been developed in the meantime for the InVeKoS System (*Integriertes Verwaltungs- und Kontroll-System*). Experiences from State Baden-Württemberg (fruit-growing in the Lake Constance region) make it clear that other administrative fields (water management, environmental protection) need to be integrated at an early phase into the concept in order for them to become willingness for participation.

The still unclarified problem of financing and evenly distributing the economical burdens is closely connected to the question of whether measures will be accepted that interfere with the land property rights of affected farmers. So far there are no experiences related to the acceptance of such measures (with the exception of "Altes Land" region). However, considering the general attitude of agricultural land owners in Germany, one can expect little willingness to accept.

9.6 Comparison of the GeoRisk approach with deterministic and scenario-based approaches

Martin Bach

Despite the fact that the authorization of pesticides is still a national legislative issue, the EU member states tried to coordinate the procedure and the recommendations on pesticide registration. Up to 2003 these activities have been organized by the FOCUS group (FOrum for Co-ordination of pesticide fate models and their USe). The FOCUS platform (http://focus.jrc.ec.europa.eu/) obtain the currently approved versions of FOCUS simulation models and FOCUS scenarios, that are used to calculate the concentrations of plant protection products in groundwater and surface water in the EU review process according to Council Directive 91/414/EEC. From the end of 2003, the EFSA (European Food Safety Authority) deals with risk assessment issues and the European Commission is responsible for the risk management decision. For a comprehensive review on state-of-the-art approaches to environmental risk assessment for pesticides see Azimonti (2006).

The parameters considered in the different approaches and their character as a deterministic or a probabilistic variable are presented in Table 9-3.

Table 9-3: Deterministic and probabilistic parameters of different risk approaches for spray drift exposure of surface waters (runoff and drainage are not regarded)

Parameter	EVA 2.0 Worst case deterministic	FOCUS Scenarios (Step 1 & 2)	GeoRisk Static exposure model (stagnant ditch)	 Dynamic exposure model (flowing river system)
Entry calculation				
Dosage	Deterministic (max recommended)			
Frequency of depositions	Deterministic (single application per field)			
Distance field – water surface	Deterministic: 3 m (or constraints: 10 or 20 m)		Geo-referenced[a] deterministic: GIS analysis (ATKIS or HR data based)	
Spray drift deposition	Probabilistic: 90^{th} percentile of the modelled spray drift deposition distributions		Geo-referenced point estimation: 90^{th} percentile from Monte Carlo-simulation of 8 wind directions[b] and BBA drift table variability	
Volatilisation deposition	Regarded		For generic substance risk assessment: not regarded For substance specific risk assessment: to be regarded (acc. To EVA)	
Drift reducing technique	Deterministic (fixed factors 50, 75, 90 %)			
Drift reducing landscape elements	Not regarded		Hedges: geo-referenced deterministic; seasonal variance 25 – 75 % River bank and river emergent vegetation: recently not regarded	
Exposure assessment				
Water body model	Deterministic: Standard ditch (static)	Deterministic: Standard ditch (static)	Deterministic: Standard ditch (static)	Geo-referenced deterministic: flowing water body (dynamic), hydrological parameters: river width, depth, flow velocity, dispersion coefficient
Deposition superposition	Not regarded		Not regarded	Geo-referenced probabilistic: distribution function[c] of superpositioning
Load calculation	(not relevant)		(not relevant)	Geo-referenced
Evaluation of ecotoxic effects				
Registration criteria	TER > Trigger value		PEC_{ini}(static) < RAC(static)	"exposure profile", maxPEC_TWA(dynamic) < RAC(dynamic)
Trait-based criteria	Not regarded		Geo-referenced (details ref. to Table 4-23, chap. 4.8)	
Risk mitigation				
Substance related measures	Application distance constraints (3, 5, 10, 20 m) Drift reducing technique (50, 75, 90 %)		Application distance constraints (3, 5, 10 m) Drift reducing technique (50, 75, 90 %)	
Landscape management	Not regarded		Drift reduction measures (options cf. Table 9-1) at water segments to be managed	

[a]) "Geo-referenced point estimate" in the context of GeoRisk means: one value per river segment (25 meter), derived from analysis of spatial data via Monte-Carlo-simulations.

[b]) Effect of wind speed on deposition is partly captured by variance of BBA drift table values.

[c]) Distribution function of the probability that a packet of running water receives 0, 1, 2, …-times a drift deposition (i.e. binomial function); the probability depends on (i) the length of a river stretch, (ii) the water needs to pass in a specified period of time, and (iii) the number of treated fields along this river stretch during the specified period of time.

Comparison of GeoRisk and FOCUS approach

Table 9-3 demonstrates that the step "entry calculation" of the FOCUS and the GeoRisk approach do not differ substantially from each other. GeoRisk introduces as new components first the deposition via volatilization for very volatile substances, and second the possibility of the presence of drift-reducing landscape elements as geo-referenced information.

The FOCUS group describes the approach as a sequence from Step 1 to Step 4. Step 1 calculations start with widely simplifying assumptions, that are progressively refined in steps 2 to 4 through the use of additional input data and model complexity (FOCUS, 2007) from headwater in the receiving water body. FOCUS Step 3 differentiates the calculation to ten scenarios, which should represent the range of soil-climate zones across Europe. Ditch, stream, and pond are considered as different water body scenarios but the scenarios cover only different situations with respect to the diluting water volume (and partly with additional pesticide loading). Finally, FOCUS Step 4 suggests adding additional realism to the many simplifying assumptions and introduces the field scale for surface water modeling. Key assumptions subject to refinement in Step 4 finally are catchment characteristics such as area, percent treated, hydrology, spatial distribution of treated fields, temporal distribution of catchment and edge-of-field loadings. For instance, Step 4 proposes to describe streams and ditches with the dynamic wave description, solving vertically integrated equations of conservation of continuity and momentum (FOCUS, 2007).

These proposals for a FOCUS Step 4 sounds similar to the approaches which have been developed in the GeoRisk project. The essential discrepancy between these two approaches is the fact, that the FOCUS proposals at the present stage are just theoretical recommendations, while GeoRisk offers solutions for the mathematical/computational description of flowing waters as a dynamic system, and the incorporation of the probabilistic elements of pesticide application pattern in time and space. Here GeoRisk has already gone ahead and has demonstrated that the probabilistic exposure assessment and risk evaluation can be put in practical application for both, static and dynamic surface water bodies.

Most enhancements affect the "exposure assessment". With the "dynamic exposure" for the first time a risk evaluation approach is presented which handles concentration profiles over time and distance in flowing waters. Therefore, this part can't be compared to the conventional stagnant ditch approach. Looking at water bodies as dynamic systems requires the introduction of a new concept of Predicted Environmental Concentration, realized in GeoRisk in the form of the one hour time weighted averaged $PEC_{\_TWA(1h)}$. Furthermore the dynamic approach needs a new concept and/or new values of the risk criteria RAC(dynamic) to compare with the $PEC_{\_TWA(1h)}$. Thus, it will take some time and (positive) experiences of the PPP community in the near future to reach full acceptance for the dynamic part of a GeoRisk registration procedure.

The process step "Evaluation of ecotoxicological effects" for the *dynamic* water body model breaks new ground in science. On the one hand, with the "$maxPEC_{TWA(1h)}$" a new criterion is introduced to judge the modeled exposure concentration. However, on the effect side, this quantity needs a new trigger value to compare it with, which can be derived from RAC for the static situation by multiplication with a factor (see Chapter 6.3) that accounts for the short-time intermittency of the pesticide exposure in a flowing watercourse. Nevertheless at the current state of debate among the stakeholders, the kind of landscape-related measures really accepted by them has not yet been finalized enough to be mentioned here to the registration authorities, official advisory boards, and the industry.

GeoRisk - some conclusions

One can generally assume that the river ecology protection target defined by the pesticide authorization procedure will be “most likely” guaranteed with the GeoRisk procedure, too. The new procedure contains without change several “conservative” parameters, i.e. distributed variables, which achieve a value during calculation that reaches the 90^{th} or higher percentile from their distribution function (for a detailed variable list cf. chapter 4.8, Table 4-23). If several of these 90-percentile values are combined (via multiplication), then there will be a very low probability that the resulting variable will reach a higher value in reality than the one calculated (the percentile of the target variable converges to 100 %). Thus a sufficiently high level of protectivity can also be awarded to the GeoRisk procedure.

The GeoRisk approach places high demands on the availability and quality of geo-data. This affects several areas: (I) ascertaining the distance between water body and pesticide treated areas; (II) capturing the presence of landscape elements (hedges, bushes, etc.) which reduce the spray drift deposition; (III) generating a consistent, topologically correct digital map of the river network; and finally (IV) assessing the hydraulic properties of the water segments (width, depth, flow velocity, dispersion coefficient).

Specifically, the hydrological attributes of all relevant water stretches in Germany cannot be conclusively derived directly from digital maps or other geo-data with current knowledge, but rather it necessitates calculations supported by models and will have to be validated via sampling (cf. Chapter 8.2).

The effort and costs of GeoRisk are higher compared to the previous approach, because first of all the dynamic approach within a GeoRisk evaluation requires elaborate calculations (especially computing time), and second of all geo-data must be procured, stored, continuously updated, and evaluated for this. According to the state of the discussion, this higher burden for the more differentiated and thereby more elaborate risk assessment would be defrayed by both industry as well as the registration authorities (at what percentage?). In light of the approval procedures, this overhead seems to be justifiable when compared to the simplification in the application of pesticides that can be achieved with this for the users.

The experiences with implementing the GeoRisk dynamic approach for the pilot area of Hallertau (rivers Lauterbach and Haunsbach, approx. 15 km of water stretches in total; cf. Chapter 8) show that the procedure is practical, and no categorical hindrances came up. Among technical aspects, the implementation for all water bodies in regions with permanent crops in Germany is primarily a question of work capacity and computing time. This especially applies when probabilistic calculations of the $maxPEC_{TWA(1h)}$ for every segment of water (which is indispensable according to the state of knowledge) is conducted with the necessary number of random distributions of spray drift depositions onto the body of water; the number of necessary Monte Carlo realizations could, for example, be prepared in the range of n=100 runs. Furthermore the uncertainties in the hydrological attributes of all relevant water segments in Germany must be captured. This uncertainty can be figured out with additional Monte Carlo runs with randomly distributed hydraulic parameters.

Concerning the feasibility or rather the effectiveness of landscape-related risk management measures, no personal experiences could be collected in regard to the project, because no measures could be implemented in practice during the project runtime. Reduction of exposure via various measures (see Chapter 9.4) is in principle indisputable and is clearly verified from pertinent literature. Assessing the effectiveness of such measures is therefore primarily a question of costs and/or of stakeholder willingness to participate in implementing it.

Annotation: The FOOTPRINT approach

Starting in 2006, a further pesticide risk evaluation approach has been developed as a EU FP-6 research project: FOOTPRINT (Functional TOOls for Pesticide RIsk assessmeNt and managemenT; http://www.eu-footprint.org/) aims at developing computer tools to evaluate the risk of pesticides impacting (via runoff and erosion, tile drainage, and spray drift) on water resources in the EU (surface water and groundwater). The tools which are being developed are designed to operate at different scales.

<u>FOOT-NES</u> (National and EU Scale): This tool allows risk assessments to be undertaken at the large scale (EU and member states scale). FOOT-NES is designed to meet the needs of EU and national policy- and decision-makers, of Environment Ministries and Agencies. The tool is also likely to be of interest to pesticide registration authorities.

<u>FOOT-CRS</u> (Catchment and Regional Scale): The tool is designed for scales ranging from small catchments to regional levels and for use by 'water managers', may they be local authorities, environment agencies, water companies or stewardship managers.

<u>FOOT-FS</u> (Farm Scale): This tool is being developed for use at the local (farm scale) by agricultural advisers and farmers.

Each of the three tools should allow: (I) the identification of pesticide contamination pathways in the landscape; (II) the estimation of levels of pesticides being transferred towards surface water and groundwater; (III) specific recommendations to be made to reduce the contamination of water resources by pesticides. All 3 tools share the same underlying principles and science and are therefore fully consistent across scales. The backbone of the risk assessment at all scales should be formed by a very huge number of model PRZM and MACRO simulation runs for all combinations of soil, climate, crop, DT_{50}, and k_{oc}, which are relevant to cover (nearly) the entire European agro-environmental scenarios. The risk assessment comprises both, deterministic and probabilistic elements. To which extend the approaches follow a “realistic worst case” philosophy or tend more to describe the “real situation” can’t be judged within the scope of the project presented here.

However, recently (status: August 2010) the FOOTPRINT tools were not fully operable and not online accessible due to a delay in the project deliverables. Therefore it is not possible to evaluate the performance of the three tools and to validate their results.

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10 Outlook - further refinements of the GeoRisk approach and application to field crops

Martin Bach

As a component of a rigorous and professionally based pesticide policy, the development of the GeoRisk approach introduced here for assessing pesticide contamination in surface water from treated crops over the spray drift path is presented only in the initial step. Further along the process, a sequence of several model developments consistent with each other will be necessary in order to be able finally to conduct the risk assessment on a geo-referenced, probabilistic basis for all crops and contamination paths. An integral part of this process will be a substantial evaluation of the quality and the uncertainty of the risk assessment tools which includes a quality check of all the parameters and variables required to run the geo-referenced PEC calculation. With this, further developments for a higher-level approach are necessary, which allow for the integration of substance degradation in water, volatilisation, as well as exchange with the sediment also for the dynamic flowing water approach. Besides this, fundamental questions about recovery and re-colonisation considerations must be clarified in the proceedings.

10.1 Implementation of the GeoRisk approach “spray drift” for field crops

It is generally impossible without considerable methodical changes to apply the GeoRisk spray drift approach (static and dynamic), described in Section 4, on field crops as well. Model construction and strategy for the exposure assessment are identical for both types of cultivated plants (field crops as well as orchards, vineyards, and hops); the necessary parameters (cf. Table 9-3) are all known. From the new data, only the spray drift deposition values for boom sprayers will have to be consulted; for surveying the geo-referenced sizes (location of treated fields, distance to surface waters, etc.), GIS analyses for the entire agricultural land with crop fields (approx. 12 million hectares) in Germany will have to be conducted. Although this would mean a significantly larger amount of work compared to that done for the acreage of permanent crops (approx. 180,000 hectares) in Germany, it generally would not be a problem.

One important difference, however, must be noted: permanent crops are perennial plants whose locations in the landscape remain (nearly) stable over long periods of time. In contrast, crop rotation is typical for agriculture, meaning that the surface area and location of potential application areas for an active substance change from year to year. The matrix of distances of the treated sites to the water system can therefore only be a probabilistic factor. The imprecision of a PRA approach such as “spray drift field crops” would consequently be significantly larger than that in permanent crops.

For a variety of reasons, the necessity of a geo-referenced PRA “spray drift field crops" is considered significantly less important in comparison to permanent crops:

- The spray drift loss of boom sprayers is approximately 3 to 10 times lower (cf. BBA drift tables) than loss of air blast sprayers (with a carrier air stream). Furthermore, permanent crops require treatment much more frequently than field crops. Both factors together lead to a situation in which spray drift entries with permanent crops, in contrast to field crops' entries over runoff and tile drainage, must be classified as dominant exposition paths for surface water in the respective regions. Bach *et al.* (2000) came to the same assessment with a model supported semi-quantitative estimate of pesticide entry in surface water over various contamination paths from the identified cultivations in Germany.

- The timeframe for treating field crops in most situations is large; the timing of herbicide treatments in the fall or spring can be stretched out over several weeks, for example. This arises out of the significantly small technical treatment capacity of agriculture in field crops, expressed in the number of field spraying devices per hundred hectares of treatment area.
- Technical efforts and overhead for drift-minimizing technology cost significantly less and are more widespread with boom sprayers using drift-reducing nozzles.
- Finally, only a relatively few number of pesticides are currently allocated for field crops with an application constraint NW 60x which bans applications closer than 15 or 20 m to surface waters. For reasons of simplification of constraints it would be an easy solution to ban all compounds with a 15/20 m constraint, but this proceeding would discriminate the producers of the respective compounds. Furthermore the main objective of a refined risk approach for field crops is not the reduction of the number of constraints but rather a more realistic risk assessment for spray drift losses from field crops.

However, for this general assessment, that spray-drift contamination in field crops is of secondary importance, a limitation must be cited. Especially in Northeast Germany, but also in other cropping regions with spacious cuts of fields, expanded field sections with the same type of crop occur, and they border several hundred meters of water. If then these fields are located close to the stretches of water and furthermore all fields are treated together within a comparably short time-frame (in the same range as the duration of the field spraying), a longer section of water could be simultaneously affected by spray drift deposition. If during a dynamic assessment this deposition only randomly encounters a “prior pollution,” meaning that the water packet is already carrying a certain concentration of substances due to one (or more) depositions in the headwaters, then theoretically a threshold value could be exceeded. Whether this scenario causes a serious risk for waters or it is only a hypothetical threat this question can not be decided in the context of the project presented here and has to be tackled by future research.

10.2 Development of a geo-referenced probabilistic risk approach for “runoff” and “tile drainage” entries from orchards, vine yards and hops

The risk of water pollution via the contamination pathways of runoff and drainage is currently assessed in the German pesticide authorization regulation on the basis of the EXPOSIT model. EXPOSIT is a conceptional, strongly simplified, deterministic calculation approach that does not enable any geo-referenced, probabilistically based exposition calculation. More differentiated model approaches for estimating pesticide runoff and drainage contamination in surface water are generally available; the two most worth mentioning are the models PRZM and MACRO. The central challenge for all such models is a sufficient, exact picture of the water flows in the ground and on its surface. The plausible description of the water flows is the indispensable requirement for properly modeling on that basis all the linked transport processes of pesticide substances.

Runoff: PRZM uses the USDA SCS curve number approach for modeling runoff. Chapter 10 of the SCS (now NRCS) National Engineering Handbook (2004) clearly states, “In flood hydrology, baseflow is generally dealt with separately, and all other types are combined into direct runoff, which consists of channel runoff, surface runoff, and subsurface flow in unknown proportions. The curve number method estimates this combined direct runoff.” The SCS curve number procedure has therefore been designed for using (smaller) catchment areas, however not for estimating the surface runoff that is the actual part of precipitation that flows on the surface of the ground. For this reason, PRZM is not recommended for a geoPRA runoff.

For measuring surface drainage (and erosion) on the field level, several models have been described in the literature, a few worth mentioning are WASim-ETH, EROSION 2/3-D (cf. DWA, 2010), and CMF (Kraft et al., 2008). These models so far are not equipped with components for illustrating pesticide transport. Frey et al. (2009) published a very promising approach for modeling the pesticide contamination of field surfaces via runoff for a small catchment area; however, it is bound to the project area and has very elaborate data. So far no systematic experiences and (independent) validations of these models from a greater number of applications exist, meaning that nothing can be stated about the quality and performance of these models during a “routine operation” for a large number of soil-climate-crop combinations. In the author's opinion, even more scientific groundwork must be developed in this area.

In reviewing the aforementioned models, one must adhere to the fact that the time and intensity of pesticide contamination in surface waters over the transport pathway runoff (and erosion) is dependent upon a number of factors; the process modeling is a lot more complex than with spray drift. All models approximating the process therefore have a greater need for entry data (data hungry); a consequence of this, a geoPRA runoff, compared to a geoPRA spray drift, will contain significantly more probabilistic elements. Because of the large number of these very spatially and temporally variable input parameters, it is to be expected that the results will be tainted with a significantly larger uncertainty.

Tile drainage: A current overview of state-of-the-art on preferential flow modeling were given by Jarvis and Dubus (2006) and others. Generally water movement in the ground, including the overflow into a drainage system as well as the connected transport of pesticides, can be better described as runoff. The model MACRO has been tested in several field experiments, which were summarily evaluated by Reichenberger (2005) with the result that the comparison of the measured and modeled findings about pesticide discharge over drainages could be rated overall as satisfactory. Basically it seems possible, therefore, to conduct a geoPRA drainage with an entry calculation based on MACRO. For a geo-referenced approach, however, there is one problem: information, specifically digital maps used for distributing drained surfaces for orchards, vineyards, and hops in Germany, is currently not normally available.

10.3 Development of a geo-referenced probabilistic risk approach for “runoff” and “tile drainage” entries from field crops

Finally, the development of a geo-referenced, probabilistic risk approach for “runoff” and “tile drainage” for entries from *field crops* (replacing EXPOSIT) would complete the set of geoPRA approaches, covering all pathways of entry as well as all kinds of agricultural land area in Germany treated with pesticides (Table 10-1). However, this last step would separately superpose all problems mentioned in Chapters 10.1 and 10.2 in the context of the respective crops and pathways of entry. Therefore this development has a long-lasting perspective.

Table 10-1: Significance of different pathways of pesticide entry into surface waters (significance in terms of estimated river load, quantification according to Bach et al. 2000, 2005), and status of geo-referenced probabilistic risk approaches for their evaluation.

Entry / Crops	Spray drift	Runoff & drainage
Orchards, vine yard, hops	Significance: high GeoRisk approach described in the report presented here	Significance: medium geoPRA to be developed (ref. 10.2)
Field crops	Significance: minor geoPRA to be developed (ref. 10.1)	Significance: very high geoPRA to be developed (ref. 10.3)

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11 Project management and workshop organisation

Roland Kubiak & Udo Hommen

In this chapter, the organisation of the project in its six workpackages and the scientists involved are shortly introduced.

Discussions between the project partners and the envolved scientists of the UBA were organized in 12 telephon and 9 face to face project meetings. In addition, two meetings of the project advisory board were organized.

Notes of the telcons and meetings as well as other relevant documents were available for the project partners and the UBA via the internet (google group).

In November 2009, the consortium organized a 3 day workshop with experts from government, industry and academia in the UBA in Dessau. The full workshop report (in German) is available as an appendix to this report. A summary is given below.

11.1 Workpackages and project partners

The project was organized in six work packages as shown in the following diagram. Work package leaders were responsible for the coordination of their work package.

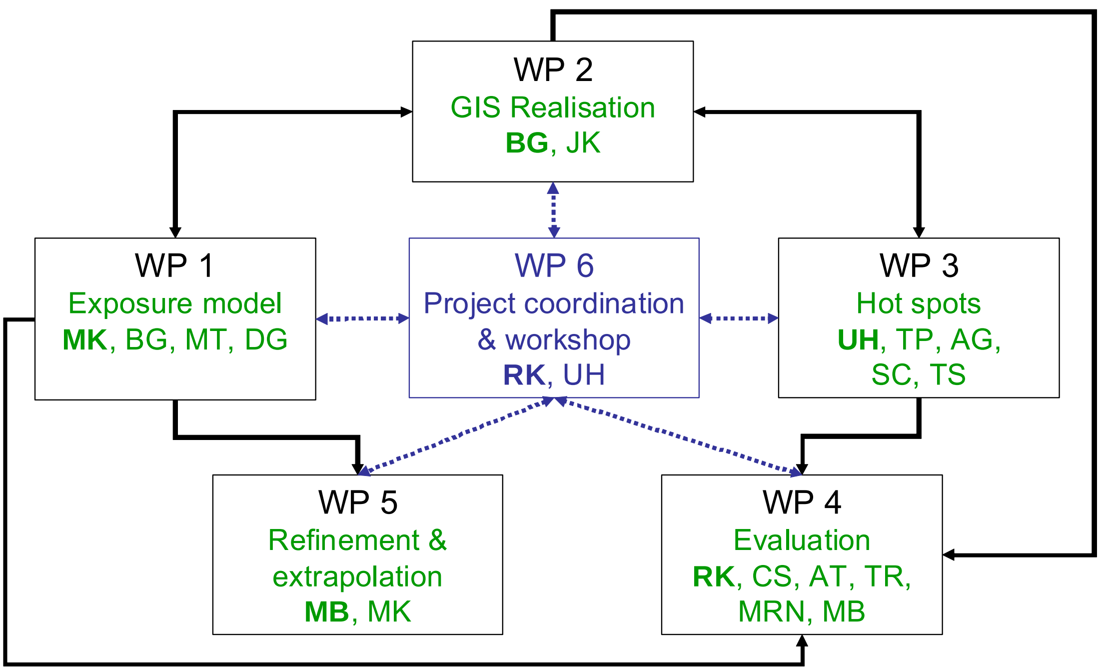

Figure 11-1: Structure of the project. Work package leaders set in bolt, abbreviations see Table 11-1

Table 11-1: Project consortium

Institution	Shortcut	Persons	Function	
RLP AgroScience	RK	Prof. Dr. Roland Kubiak	Head of consortium	Head WP 4 & 6
	MT	Dr. Matthias Trapp		
	DG	Djamal Guerniche		
Fraunhofer IME	UH	Dr. Udo Hommen	Deputy	Head WP 3
	MK	Dr. Michael Klein		Head WP 1
	CS	Dr. Christoph Schäfers		
JKI	BG	Dr. Burkard Golla		Head WP 2
	JG	Jens Krumpe		
IER, RWTH Aachen	TR	Prof. Dr. Toni Ratte		
	TP	Dr. Thomas Preuß		
	MRN	Dr. Martina Roß-Nickoll		
	AT	Dr. Andreas Toschki		
	AG	Dipl. Biol. Andre Gergs		
gaiac, Aachen	SC	Dipl. Biol. Silke Claßen		
	TS	Dr. Tido Strauss		
ILR, Univ. Gießen	MB	Dr. Martin Bach		Head WP 5

Alexandra Müller and Steffen Matezki were the responsible scientific contacts in the UBA. In addition, Dr. Jörn Wogram, Dr. Anne Osterwald, and Dr. Christina Pickl participated in telcons and project meetings.

11.2 Telephone conferences

Number	Date	Major Tasks
1	27.02.2008	Preparation of 1st Project meeting
2	28.04.2008	Organisation and to do´s from the Dessau meeting
3	06.06.2008	Discussion on a german-wide PEC and the hotspot definition
4	20.10.2008	Tasks of WP 4
5	24.10.2008	Approach of modelling streams
6	26.11.2008	Polygons and lines
7	26.01.2009	Tasks of AP 1
8	10.02.2009	Work still to finalise in WP 1
9	28.04.2009	Update of all WP
10	26.05.2009	Finalisation of Aachen meeting protocols and work on to do list
11	21.12.2009	Discussion on UBA comments on the intermediate report
12	29.03.2010	Results of dynamic PEC calculation; organisation of final re-port

11.3 Project meetings

Number	Date	Place	Major Tasks
1	27.3.2008	RLP AgroScience Neustadt	WP 1 – 3
2	7.4.2008	UBA, Dessau	Clarification on project aims
3	3.7.2008	UBA, Dessau	General agreements on tasks and aims
4	1. – 2.9.2008	RLP AgroScience Neustadt	Detailed work on WP 1 - 3
5	15. – 16.12.2008	UBA, Dessau	Final discussion on AP 1, discussion on AP 3
6	4. – 5.5.2009	TH Aachen	Preparations on intermediate report
7	29.9.2009	Uni Giessen	
8	27. – 28.4.2010	Fraunhofer IME Schmallenberg	Final report structure and implemen-tation
9	6.7. – 7.7.2010	UBA, Dessau	Discussion of draft report

11.4 Advisory Board meetings

The advisory board members were:

Name	Organisation	e-mail
Dr. Sabine Gärtner	BMU	sabine.gaertner@ bmu.bund.de
Eva Dressler	BMU	eva.dressler@bmu.bund.de
Dr. Manfred Klein	BfN	manfred.klein@bfn.de
Dr. Friedrich Dechet	IVA	dechet.iva@vci.de
Thorsten Schad	for IVA	thorsten.schad@bayercropscience.com
Michael Morgenstern	LVLF Brandenb.	Michael.Morgenstern@LVLF.Brandenburg.de
Christian Feld	Uni Duisburg	christian.feld@uni-due.de
Dr. Martin Streloke	BVL	martin.streloke@bvl.bund.de
Dr. Matthias Liess	UFZ	matthias.liess@ufz.de
Dr. Michael Glas	PS - BW	michael.glas@ltz.bwl.de

Until now (August 2010), two advisory board meetings have taken place:

1. December 12, 2008 in Dessau
2. September 30, 2009 in Giessen

11.5 Workshop

A three day workshop with in total 34 participants from academia, government / regulation, and industry was held from 16. – 18. November 2009 at the Umweltbundesamt in Dessau. The full workshop report (in German) is available as appendix C of this report and thus, only a summary is given here:

The current state of the GeoRisk Project (UBA R+D Project 3707 63 4001) including options and difficulties regarding the implementation of results was discussed in November 2009 with representatives of various stakeholders. The discussions focused on the evaluation of risks caused by drift entry from orchard-, wine- and hop cultures.

The following results from the GeoRisk Project were presented:

– Nationwide calculation of the PEC_{sw} (Predicted Environmental Concentration in surface water) on the basis of ATKIS and "standard ditch model": geo-referenced probabilistic calculation of the entries and the expected km management segments per culture in Germany
– Differences between the PECs based on ATKIS and High Resolution Data (HR are greater in heterogeneous small-scale structured landscapes than in homogeneous landscapes)
– Method for the generation of a topologically correct, directed network of water bodies which is a prerequisite for a dynamic modelling of streams
– Investigation of the sensitive parameters for the derivation of PECs in streams

- Calculation of concentration curves for selected waters under conservative assumptions
- Elaboration of ecotoxicological hotspot criteria related to trait groups (under development)
- Concept for the evaluation of ecotoxicological effects caused by short-term exposure (shorter than in standard tests)
- Prototype of a web-based application tool for the calculation of geo-referenced PECs and for identifying hotspots (spatial accumulation of critical loads)

The main statements developed in the work groups and plenary discussions of the workshop were as follows:

- The objective of a geo-referenced and probabilistic procedure for the risk assessment of pesticides in Germany is a most realistic risk assessment which should also provide the basis for simplifications and facilitations of the current nationwide substance-related regulations of pesticide application ensuring at the same time a sufficient protection level
- The target of protection "no unacceptable effects on local populations" should be reached with 95 % certainty, considering that comparable estimations usually include considerable uncertainties. This does not mean that also a 95th centile of a local entry or PEC-distribution must be used, but that unacceptable effects on local populations can be excluded with a certainty of 95 % when a pesticide is applied under consideration of the respective risk management requirements. A 90th centile of the local entry or PEC distribution, for example, can also contribute to the desired certainty of the total assessment procedure.
- Uncertainties in the geological data (errors of measurement, classification errors etc) have to be minimized; procedural uncertainties have to be analysed systematically (analysis of uncertainty) and relevant parameters taking influence on the risk assessment (sensitivity analysis) need to be pointed out.
- The calculation of initial PECs on the basis of the standard ditch model (stagnant ditch, 30 m depth) is to be considered as a geo-referenced probabilistic estimation of the drift entry. For a real geo-referenced probabilistic estimation of exposure the presented approach of a dynamic model allowing the inclusion of realistic hydrodynamic data (e.g. width-depth ratio, flow velocity) as well as transport and dispersion in streams is considered suitable. Before the model can be applied nationwide a considerable working and development effort will be necessary.
- A (theoretical) case study showed that decisive parameters taking influence on exposure estimation in the context of a dynamic water model are the probability of applications and the time window of applications in the upper course of a considered segment. Application over two days was considered a preliminary plausible assumption; for a more detailed clarification the NEPTUN data could be useful.
- With respect to a proposed equal distribution of wind directions during the application and the possibility of identifying preferred local wind directions in a drift-relevant height over the ground it was stated that the realistic distribution of the wind directions is of no importance for the PEC distribution at the considered site when using a high percentile, as e.g. the 90th percentile.
- The participants agreed in that it has to be differentiated between a generic, substance-independent analysis and the (substance specific) registration practise later on.

- The generic risk assessment should include a so-called hotspot analysis for the identification of water segments with an ecologically critical accumulation of risk segments under consideration of the recovery and resettlement potential of relevant populations.
  The preliminary basis for the hotspot criteria is an approach proposed by the German Federal Environment Agency (UBA, 2007): The regulatory acceptable concentration (RAC, derived from ecotoxicological testing) should be exceeded by the predicted concentration (PEC) in not more than 10 % of a 1000 m water segment.
- The project intends to refine this conservative criterion in so far that dose-effect relationships, shorter exposure periods than applied in the ecotoxicological tests as well as the different levels of tolerable effects relevant for the different species can be considered (trait approach).
- The consideration of hedges and bankside herb vegetation as drift-reducing factors can be performed for specific cultures also in the generic analysis; in this case it must be guaranteed that pesticides will be applied only in seasons when the hedges are fully foliated or when the weedage of the water bodies is strong. Further refinement is substance-dependent (e.g. consideration of the fate of the substance in the water body).
- The generic risk analysis is the basis for a landscape-related risk management in identifying water segments for which ecologically critical entries are expected which can be reduced by management measures.
- Manner and extent of a possible risk management in hotspots still needs to be clarified.
- There was no agreement on whether the development of (new) hotspots should present the subsequent registration criterion, as proposed in the GeoRisk approach. The IVA (Industrieverband Agrar, industrial association for agriculture) proposed to develop the generic hotspot analysis and the resulting generic risk management as proposed in GeoRisk, but the decision upon registration in no case should be based on the presence of single hotspots. BVL (Bundesamt für Verbraucherschutz und Lebensmittelsicherheit, Federal Office of Consumer Protection and Food Safety) and IVA proposed to draw the decision on the registration of pesticides on the basis of nationwide PEC-distribution over all (relevant) water segments and not to apply any longer the „edge of the field“-approach.
- Agreement was achieved neither on the required steps to elaborate necessary geodata including central data management nor for the elaboration and positioning of a tool (model) for the exposure and risk assessment.
  GeoRisk proposes a central tool: The administrative body (UBA) will establish a geodata and exposure data bank which will be updated in certain intervals. A web-based tool allows the input of substance-dependent data on application and toxicity. A refinement of the geological data in the course of one registration procedure is considered impracticable by the UBA and therefore is not intended. A refinement of geological data can only be carried out in the scope of the cyclic record updating; it will then be accessible for all applicants.
- The project provides progress in knowledge which should be considered according to the federal plant protection act. The implementation of a new procedure considering the dynamic stream model approach including an adapted evaluation of short-term exposure is principally possible and is preferred to a static ditch model by the workshop participants.
  A nationwide application, however, still requires considerable improvement and work steps.

Following the dynamic stream model and according to estimations based on the case study the number of management segments (MS) is significantly below the number determined with the statistic ditch model. However, detailed statements about the extent of risk management in hotspots are not yet possible so far.

- The participants of the workshop principally agreed on the next steps required:
    1. Finalisation of the present project
    2. Joint elaboration of the parameters required for the determination of the management segments (MS) based on the dynamic stream model
    3. Calculation of the MS for a pilot area (e.g. the Hallertau) and, if possible, at least one further area with a different crop
    4. Decision about the establishment of a pilot project on this basis
    5. Performance of a pilot project, e.g. a field test in the Hallertau as a clearly defined area characterized by one specific culture, including a chemical and biological monitoring
    6. Decision about the implementation of the procedure starting with all orchard-, wine- and hop cultures
    7. In general there was a demand for an active coordination point for the implementation of a geo-referenced probabilistic approach, e.g. by the BVL steering committee "Probabilistik".

11.6 Projects reports, poster and platform presentations, publications

11.6.1 Project reports

Short project report:	September 2008
Intermediate Report:	July 2009
Draft report 1	June 2010
Draft report 2	31 August 2010
Draft report 3	03 December 2010

11.6.2 Platform presentations

Bach, M.; Trapp, M.; Guerniche, D. (2010): GeoRisk: Grundlagen der PEC-Berechnung. Deutsche Pflanzenschutztagung, Berlin, Sept. 2010

Golla, B., Krumpe, J., Strassemeyer, J., Gutsche, V. (2009): Spatio-temporal network analysis - An approach for the exposure assessment of small streams. SETAC Europe Meeting, Göteborg, Sweden

Hommen, U. et al. (2010): GeoRisk: Further advancements in the development of a spatial approach for the assessment and management of environmental risks of plant protection products in Germany. SETAC EU Annual Meeting May 2010, Sevilla

Hommen, U., Gergs, A., Preuss, T.G. (2010): Trait based population models to estimate tolerable effect levels in a geo-data based probabilistic risk assessment of pesticides SETAC NA, November 9th, 2010

Hommen, U., Kubiak, R., Bach, M., Golla, B., Klein, M., Matezki, S., Müller, A. (2010): GeoRisk: ein georeferenzierter probabilistischer Ansatz zur Risikobewertung von Drifteinträgen in Oberflächengewässer. Vortrag. Deutsche Pflanzenschutztagung, Berlin, Sept. 2010

Hommen; U. et al. (2009): Das GeoRisk-Projekt: Ein neuer Ansatz zur georeferenzierten probabilistischen Risikobewertung von Pflanzenschutzmitteln. Workshop Aquatische Expositionsszenarien in der Zulassung von PSM – Internationale Entwicklung und Optionen für die Schweiz. Wädenswil, Switzerland, 20.10.2009

Krumpe J, Golla B (2009): Multikriterielle Analyse verteilter Geoinformationen am Beispiel eines webbasierten Entscheidungsunterstützenden Systems zur Risikoabschätzung von Pflanzenschutzmitteln- Geoinformatik 2009

Krumpe J, Golla B (2009): Multiattributive Raumbewertung von verteilten Geoinformationen. Vortrag. Degree-day 27.05.2009

Kubiak, R. et al. (2009): A new spatial approach for the assessment and management of environmental risks of plant protection products in Germany. SETAC EU Annual Meeting May 2009, Göteborg

Preuss, T.G., Gergs, A., Classen, S., Strauß, T., Ratte, H.T., Hommen, U. (2010): GeoRISK: Ökologische Kriterien als Basis für die georeferenzierte Risikoabschätzung von Pflanzenschutzmitteln in Oberflächengewässern. Vortrag. Deutsche Pflanzenschutztagung, Berlin, Sept. 2010

Trapp, M.; Guerniche, D.; Bach, M.; Kubiak, R. (2010): GeoRisk: Raumzeitliche Simulation von PEC in Fließgewässern (Beispiel Hallertau). Vortrag. Deutsche Pflanzenschutztagung, Berlin, Sept. 2010

Trapp, M., Guerniche, D., Bach, M., Trapp, Kubiak, R. (2010): Pesticide exposure assessment in flowing waters – Results for Predicted Environmental Concentrations in some brooks in Germany. SETAC EU Annual Meeting May 2010, Sevilla

11.6.3 Poster presentations

Gergs, A., Classen, S., Strauss, T., Ratte, H.T., Preuss, T.G., Hommen, U. (2009): Recovery of aquatic ecosystems in the environment – a literature review. SETAC Europe Meeting, Göteborg, Sweden

Golla, B., Krumpe, J., Klein, M. (2009): Model and parameters for a geo-referenced probabilistic drift entry calculations for permanent crops in Germany. SETAC EU Annual Meeting May 2010, Sevilla, Spain.

Gergs, A., Classen, S., Preuss, T.G., Hommen, U. (2010): Using a trait based approach to define representative species for spatial explicit aquatic risk assessment of pesticides. SETAC EU Annual Meeting May 2010, Sevilla, Spain.

Elbers, S., Schapke, J., Trapp, M., Hommen, U., Preuss, T.G., Ratte, T., Classen, S. (2010): Effects of pesticides on freshwater communities – An example from an intense hops agricultural region in Germany. SETAC EU Annual Meeting May 2010, Sevilla, Spain.

Hommen, U., Classen, S., Gergs, A., Preuss, T.G. (2010): How to assess spatial aggregation of indicated risks due to pesticide entries in water sheds. SETAC EU Annual Meeting May 2010, Sevilla, Spain.

Schaefers, C., Preuss, T.G., Hommen, U. (2010): A pragmatic but protective approach to estimate effects of pesticide ulse exposure in aquatic ecosystems. SETAC EU Annual Meeting, May 2010, Sevilla, Spain.

11.6.4 Publications

Elbers S. (2011): Analyse des Einflusses anthropogener Faktoren auf die Lebensgemeinschaft des Makrozoobenthos in Gräben einer intensiv genutzten Agrarlandschaft am Beispiel der Hallertau. Master thesis, RWTH Aachen University, Germany

Gergs A, Classen S, Hommen U, Preuss TG. 2011- Identification of realistic worst case aquatic macroinvertebrate species for prospective risk assessment using the trait concept. ESPR 18:1316–1323

Krumpe, J (2010): Multikriterielle Analyse verteilter Geoinformationen am Beispiel eines webbasierten entscheidungsunter-stützenden Systems zur Risikoabschätzung von Pflanzenschutzmitteln. GIS.Science 2/2010, 74-82l

Krumpe J: (2009): Multiattributive Raumbewertung mit verteilten Geodaten - Für ein entscheidungsunterstützendes System in der Umweltrisikobewertung. Master thesis. Zentrum für GeoInformatik (Z_GIS) der Paris Lodron-Universität Salzburg. http://www.unigis.ac.at/club/bibliothek/pdf/1346.pdf

12 Glossar

AMA (Schulz et al. 2007[51])
active risk management section = hotspot within this report
Aktiver Risikomanagement-Abschnitt: Gewässerabschnitt, in dem aufgrund einer räumlichen Aggregation von Segmenten mit vorhergesagter Umweltkonzentration (PEC) > $RAC_{realistic}$ worst case unvertretbare Effekte auf die Populationen von Nichtzielorganismen nicht mit hinreichender Sicherheit ausgeschlossen werden können. Ein AMA kann neben den gehäuft auftretenden Segmenten mit PEC > $RAC_{realistic\ worst\ case}$ somit auch Segmente enthalten, in denen eine Überschreitung der $RAC_{realistic\ worst\ case}$ mit hinreichender Sicherheit ausgeschlossen werden kann. Die erforderlichen Risikomanagementmaßnahmen im AMA beziehen sich selbstverständlich nur auf die Segmente mit zu erwartender PEC > $RAC_{realistic\ worst\ case}$, d.h. für die Umsetzung von lokalen Managementmaßnahmen ist nur die Markierung der Segmente mit zu erwartender PEC > $RAC_{realistic\ worst\ case}$ innerhalb eines AMA erforderlich (UBA 2007, unpublished)

ATKIS (Schulz et al. 2007)
acronym for Amtliches Topographisch-Kartographische Informationssystem, officail topographical geoinformation system in Germany
ATKIS steht als Akronym für das zum Zwecke der digitalen Führung der Ergebnisse der topographischen Landesaufnahme und der amtlichen topographischen Karten auf Empfehlung der Arbeitsgemeinschaft der Vermessungsverwaltungen der Länder der Bundesrepublik Deutschland (AdV) von den Landesvermessungsämtern und dem Bundesamt für Kartographie und Geodäsie (BKG) seit 1990 aufgebaut wird.

Centile (Hart et al. 2006[52])
Same as quantile, but with the proportion expressed as a percentage. The median is the 50^{th} centile.
Das gleche wie eine Quantile, aber als Prozent ausgedrückt. Der Median entsprict dem 50. Zentile

Cumulative Distribution Function (CDF) (Hart et al. 2006)
A function expressing the probability that a random variable is less than or equal to a certain value. The CDF is obtained by integration of the PDF for a continuous random variable, or summation of the PDF in the case of a discrete random variable.
Eine Funktion, die die Wahrscheinlichkeit beschreibt, dass eine Zufallsvariable kleiner oder gleich einem bestimmten Wert ist. Die CDF ergibst sich aus dem Integtral der PDF (Probability Density Function) bei diskreten Zufallsvariablen.

[51] Schulz, R., Elsaesser, D., Ohliger, R., Stehle, S., Zenker, K. (2007): Umsetzung der georeferenzierten probabilistischen Risikobewertung in den Vollzug des PflSchG – Pilotphase – Dauerkulturen. Endbericht zum F & E- Vorhaben 206 63 402 des Umweltbundesamtes, Institut für Umweltwissenschaften, Universität Koblenz-Landau, Landau, Germany, 129 p.

[52] Hart A et al. (2006): EUFRAM - Concerted action to develop a European Framework for probabilistic risk assessment of the environmental impacts of pesticides. Contract Number QLK5 - CT 2002 01346. www.eufram.com

Digital Landscape Model (DLM)
DLM describe the topographic objects of a landscape and the relief by vectors
Digitale Landschaftsmodelle (Schulz et al. 2007): Digitale Landschaftsmodelle beschreiben die topographischen Objekte der Landschaft und das Relief der Erdoberfläche im Vektorformat.

Drift
a) loss of the plant protection product during the spraying process due to wind
b) dispersal of stream / river organisms by the flow
a) Verlust von Pflanzenschutzmitte währedn der Sprühapplikation durch Wind (auch Abdrift)
b) Verbreitung von Organismen mit der Strömung

Dynamic approach
Within GeoRisk the dynamic approach means the calculation of PEC dynamics in lotic waters resulting from entries upstream. Within this approach, hdydrodynamic geo-referenced data such as water depth, flow direction and flow rates are used to consider transport and dilution of pesticides in lotic waters. The timing of applications to the single fields is explicitely modelled. Local PEC profiles are summarized a time weighted average concentrations, e.g. the maximum $PEC_{TWA(1h)}$, and Time over Threshold (ToTh).
In GeoRisk wird mit dem dynamischen Ansatz die Berechnung von PEC-Zeitreihen inFließgewässer als Ergebnis der Einträge von Pflanzenschutzmittel stromaufwärts verstanden. Der Ansatz nutzt georeferenzierte hdydrodynamische Daten wie Wassertiefe, Fließrichtung, Abflussraten zur Berücksichtigung von Transport und Verdünnungsprozessen in Fließgewässern. Das zeitliche Muster der Pflanzenschutzmittelanwendungen im Oberlauf wird dabei explizit berücksichtigt. Dielokalen PEC-Profile werden mit zeitlich gewichteten Mittelwerten, z.B. dem maximalen $PEC_{TWA(1h)}$, und Zeiten, in denen der PEC über einem Schwellenwert liegt (Time over Threshold), zusammengefasst.

Ecotoxicologically Relevant Concentration (ERC)
Boesten et al. (2007[53]): "A crucial step is to define which type of field concentration is needed as the exposure input to the effect tiers. The choice should be based on ecotoxicological considerations because this should be the concentration that gives the best correlation to ecotoxicological effects. This type of concentration is defined here as the ‘ecotoxicologically relevant concentration’ (abbreviated to ‘ERC’). The ecotoxicological considerations determining the ERC may include: (i) in which environmental compartment do the organisms live (e.g., in water and sediment)? (ii) what is the mode of action of the pesticide? (iii) what is bioavailable for the organism? (iv) what is the influence of the exposure pattern (e.g., short peaks or constant concentration over long periods) on the type and degree of effects? and (v) was the whole test duration of an ecotoxicological study necessary to cause the measured effects or would a shorter exposure period have given the same effect? It is of course necessary that the ERC is based on information available in the first tier of the effect assessment. ...
For instance, for aquatic organisms the ERC could be e.g., the maximum over time or some

[53] Boesten JJTI, Köpp H, Adriaanse PI, Brock TCM, Forbes VE (2007): Conceptual model for improving the link between exposure and effects in the aquatic risk assessment of pesticides. Ecotoxicology and Environmental Safety 66 (2007) 291–308

time-weighted average of the concentration of dissolved pesticide in surface water. For sediment-dwelling organisms that live predominantly in the top centimetres of sediment, the ERC could be the maximum over time of the pore water concentration in the top 2 cm of the sediment (or as an alternative to the pore water concentration the bulk sediment concentration)."
Ein Typ von Konzentration, in dem PEC und RAC ausgedrückt werden müssen, um verglichen werden zu können (z.B: Peak-Konzentration, TWA über 7 Tage)

Emerse vegetation (Schulz et al. 2007)
Vegetation above the water surface which mitght serve as a shield against drift entries, includes emerse aquatica macrophytes as well vegetation close to the wate
Jegliche Vegetation, die sich über der Wasseroberfläche befindet und somit potentiell PSM-Einträge über Driftdeposition nach Abdrift abschirmen könnte. Hierzu können zum einen die emersen Teile von Wasserpflanzen (z.B. Schilf), aber auch Ufervegetation zählen, die sich über die Wasseroberfläche erstreckt (z.B. Brombeeren).

Frequency distribution
The organization of data to show how often certain values or ranges of values occur (Hart et al. 2006)
Methode zur statistischen Beschreibung von Daten (Messwerten, Merkmalswerten). Mathematisch gesehen ist eine Häufigkeitsverteilung eine Funktion, die zu jedem vorgekommenen Wert angibt, wie häufig dieser Wert vorgekommen ist. Man kann eine solche Verteilung als Tabelle, als Grafik oder modellhaft über eine Funktionsgleichung beschreiben. (www.wikipedial.de)

Geographic Information Systems GIS
Systems that allow the interrelation of quality data (as well as other information) from a diversity of sources based on multi-layered geographical information processing techniques (Hart et al. 2006).
Informationssysteme zur Erfassung, Bearbeitung, Organisation, Analyse und Präsentation geografischer Daten. Geoinformationssysteme umfassen die dazu benötigte Hardware, Software, Daten und Anwendungen (www.wikipedia.de)

Generic (risk) assessment
A risk assessment based on a realistic worst case substance, i.e. a substance where the RAC is just equal the PEC resulting from the maximum mitigation measure. In GeoRisk, generic assessment have been conducted under the assumption that the substance is applied with 1 kg a.s./ha and that its RAC is equal to the PEC resulting from dritft entries from 20 m distance.
Eine Risikoabschätzung für eine virtuelle worst case substanz, die bei maximalen Abstandsauflagen gerade noch zulassungsfähig ist. In GeoRisk wurden generische Analysen unter der Annahmen durchgeführt, dass die Substanz mit 1 kg a.s./ha angewendet wird und dass die RAC der PEC entspricht, die sich für Drifteinträge bei einer Abstandsauflage von 20 m ergibt.

Higher Tier Assessment
Possible steps in the risk assessment going beyound the standard approaches in order to refine the assessment by more realistic or less uncertain data. Examples are additional single species tests, tests under a more realistic exposure situation, or micro-/mesocosm studies.
Schritte in der Risikoabschätzung, die über die Standardanforderungen hinausgehen. Beispiele sind Tests mit Nicht-Standard-Arten, Tests mit realistischer Exposition, Mikro-/Mesokosmosstudien.

Hotspot
an ecological critical aggregation of risk segments
eine ökologisch kritische räumliche Häufung von Risikosegmenten. Schulz et al. 2007: Gewässerabschnitt, in dem die im Pflanzenschutzgesetz und der EU Direktive 91/414/EEC genannten Schutzziele nicht mit ausreichend hoher Sicherheit eingehalten werden. In diesen Segmenten oder Abschnitten besteht daher ein erhöhtes Risiko des Auftretens populationsrelevanter Auswirkungen, so dass diese Gewässerbereiche spezifische, standortbezogene, aktive => Risikomanagementmaßnahmen erfordern.

Hotspot criterion
A criterion defining the spatial scale and the related acceptable effects to define a hotspot. The generic hotspot criterio proposed by GeoRisk is: Along 1000 m water body no more than 10 % effect based on the RAC and the PEC values estimated for the water body segments.
Ein Kriterion um Hotspots zu identifiizieren. Das generische Hotspotkriterium nach GeoRisk ist: Auf 1000 m Gewässerstrecke nich mehr als 10 % Effekt auf der Basis der RAC und der PEC-Werte für die einzelen Segmente.

Landscape level distribution
Frequency distribution describing the spatial variability of a property on the landscape level. In the context of geodata based probabilistic risk assessment, often a distribution based on the PECs for all relevant water bodies of a region, as state or country. From each segment a specific centile of the local PEC distribution is used to built the landscape level distribution.
Eine Häufigkeitsverteilung, die die räumlich Variabilität einer Größe für eine Region, ein Bundesland, den Staat oder eine andere räumliche Einheit widerspiegelt. Im Zusammenhang mit der georeferenzierten probabilistischen Risikobewertung von Pflanzenschutzmitteln, oft eine Verteilung der für PECs für die einzelen Gewässersegmente. Für jedes Segment wird aus der lokalen Verteilung der PEC-Werte dort eine bestimmte Zentile für die Landschaftsverteilung verwendet.

Local distribution
A frequency distribution based on Monte-Carlo-Simulations calculation the entries into or the resulting PECs in one single water body segment. This distribution describes the variability of entry envents.
Eine Häufigkeitsverteilung der Einträge oder PECs einem einzeln Gewässersegment. Diese Verteilung beschreibt die Variabilität der Einträgsereignisse.

Low bank vegetation / Niedrige Ufervegetation (Schulz et al. 2007)
Vegetation along the banks which can reduce the drift entries into the water body, e.g. herbs, grass.
Vegetation im Uferbereich, deren Oberfläche die verdrifteten Pflanzenschutzmittel abfangen, (z.B. krautige Vegetation).

Monte Carlo simulation
A resampling technique frequently used in uncertainty analysis in risk assessments to estimate the distribution of a model's output parameter (Hart et al. 2006).
Ein Verfahren aus der Stochastik, bei dem sehr häufig durchgeführte Zufallsexperimente die Basis darstellen. Es wird dabei versucht, mit Hilfe der Wahrscheinlichkeitstheorie analytisch nicht oder nur aufwändig lösbare Probleme numerisch zu lösen. Als Grundlage ist vor allem das Gesetz der großen Zahlen zu sehen. Die Zufallsexperimente können entweder – etwa durch

Würfeln – real durchgeführt werden oder durch Erzeugung von geeigneten Zufallszahlen. Computergenerierte Vorgänge können den Prozess in ausreichend häufigen Zufallsereignissen simulieren. (www.wikipedia.de)

Moving Windows
Approach to identify hotspots by moving a virtual window a given length (usually 1000 m in the GeoRiskl applications) along the water bodies to assess the proportion of segments with PEC > RAC
Verfahren bei der Identifizierung von hotspots, bei dem ein Fenster einer bestimmten Größe (z.B. 1000 m) entlang eines Gewässers bewegt wird, um den Anteil von Segmenten mit RAC-Überschreitung zu erfassen.

PEC
Predicted Environmental Concentration: The concentration of a substance in the environment that is predicted from its properties, its use and discharge patterns, and properties of the environment
Vorhergesagte Konzentration eines Stoffes in der Umwelt

PEC_{ini}
initial PEC due to an entry into a lentic water body
initiale PEC durch den Eintrag in ein stehendes Gewässer

Percentile
Incorrect term for centile used by people with IQs less than 240 (Hart et al. 2006).
Das x. Perzentil bezeichnet den Wert einer Variablen, der einer kumulativen relativen Häufigkeit bzw. kumulativen Wahrscheinlichkeit von X % entspricht. Das 90. Perzentil einer PEC-Häufigkeitsverteilung ist z.B. die Konzentration, die nur in 10 % der Fälle überschritten wurde.

Probability (Hart et al. 2006)
According to the frequentist view, the probability is the frequency of an event in an infinite repetition of identical and independent trials. In the Bayesian view, probability is a measure for the degree of belief in possible values of a random variable. In both views, probability is a measure of uncertainty of some outcome of an experiment, extrapolation, or a prediction.
Wahrscheinlichkeit: relative Häufigkeit eines Ereignisses bei unendlich vielen Wiederholungen. Maß der Unsicherheit über das Eintreffen eines Ereignisses

Probabilistic Risk Assessment (Hart et al. 2006)
Risk assessment where the probability or likelihood of adverse effects is estimated from more than one datum and the uncertainty is characterized. Within GeoRisk, the wind direction at the time of application, the variability of the deposition rate, and the temporal pattern of application events are treated a stochastic events and thus, are considered by a probabilistic approach.
Eine Risikoabschätzung, bei der die Wahrscheinlichkeit von schädlichen Effekten von aus mehr als einem Wert abgeschätzt wird und bei der die Unsicherheit charakerisiert wird. In GeoRisk werden die Windrichtung zum Zeitpunkt der Applikation, die Variabilität der Depositionsrate und das zeitliche Muster der Applikationsereignisse probabilistisch behandelt.

Probability Density Function PDF (Hart et al. 2006)
For a continuous random variable, the PDF expresses the probability that the random variable belongs to some very small interval. For a discrete random variable, the PDF expresses the probability that the random variable is equal to a specific (discrete) value.

Gibt an, wie sich die Wahrscheinlichkeiten auf die möglichen Zufallsergebnisse, insbesondere die möglichen Werte einer Zufallsvariable, verteilen (www.wikipedia.com).

Quantile (Hart et al. 2006)
The value of a random variable that corresponds to a specified proportion of the PDF of that random variable. Quantiles can be determined from the inverse CDF. The median is the 0.5th quantile. The quartiles are the 0.25th, 0.50th, and 0.75th quantiles. The centiles the 0.01th, 0.02th, … quantiles.

Wert einer Zufallsvariable, die einem bestimmten Anteil der PDF entspricht.
Ein p-Quantil ist ein Lagemaß in der Statistik, wobei p eine reelle Zahl zwischen 0 und 1 ist. Das p-Quantil ist ein Merkmalswert, der die Verteilung einer Variablen bzw. Zufallsvariablen in zwei Abschnitte unterteilt. Links vom p-Quantil liegen 100 p Prozent aller Beobachtungswerte bzw. 100 p Prozent der Gesamtzahl der Zufallswerte. Rechts davon liegen 100 (1-p) Prozent aller Beobachtungswerte bzw. 100 (1-p) Prozent der Gesamtzahl der Zufallswerte.
(www.wikipedia.de)

RAC - Regulatory Acceptable Concentration
Ecological threshold concentration, derived from ecotoxicological data sets by apply safety (or assessment) factor to the assessment endpoint of a study.
"Within an effect tier, the measured NOEC, EC_{50} or NOEAEC may not always be the assessment endpoint because it may have to be multiplied with a certain safety factor (see e.g., TER values of 10 and 100 used by European Commission (2002a), and an example of EFSA, 2005) or extrapolated with a certain model (e.g., HC_5 calculations). We assume here that the assessment endpoint of any effect tier can be simply called the 'regulatory acceptable concentration (RAC)' level thus including already any safety factors or extrapolation methods that are considered necessary. Once this RAC level has been determined, it has to be compared with the endpoint of an exposure tier (i.e., the field concentration level, called PEC level) after which it can be decided whether the risk according to this tier is acceptable." (Boesten et al. 2007)
Effektschwellenkonzentration, die aus ökotoxikologischen Studien unter Anwendung von Sicherheitsfaktoren abgeleitet wird. Früher auch EAC (Ecologically Acceptable Conc) genannt.
Im Rahmen der Pflanzenschutzmittelzulassung nach Dir 91/414 EEC entspricht die RAC den Toxizitätswerten geteilt durch die in den Uniform Priniciples niedergelegten Triggerwerten (oder den für Higher Tier Studien verwendeten Triggern). Also z.B. einer Fisch LC50 / 1000, einer Algen EC50 /10, einer Daphnien NOEC / 10.

Risk (Hart et al. 2006)
The predicted or actual probability of occurrence of an adverse effect on humans or the environment as a result of exposure to a stressor or mixture of stressors.
Die vorhergesagte oder aktuelle Wahrscheinlichkeit des Eintretens eines schädlichen Effekts auf Menschen oder die Umwelt als Folge der Exposition zu einem Stressor oder einer Mischung von Stressoren

Risk managament
here: any option to reduce the pesticide entry into surface water, these options can be related to the product (e.g. keeping minimum distances of application to surface waters, use of drift redcing application techniques), or can be local actions (e.g. planting of hedges,management of bank vegetation, use of drift mitigating fences etc.)
Jede Form von Maßnahmen, die zur Produktion des potentiellen von PSM ausgehenden Risi-

kos auf Nichtzielökosysteme getroffen werden. Hierzu zählen neben eher passiven regulativen Aspekten (z.B. Abstandsauflagen) auch aktive Maßnahmen (z.B. Anpflanzen von Hecken) (=> hotspots) (Schulz et al. 2007)

Risk segment
A water body segment where the predicted exposure (PEC) exceeds the Regulatory Acceptable Concentration (RAC)
Ein Gewässersegment, in dem die PEC die RAC überschreitet

Segment (water body segment)
smallest unit for which a PEC estimation is conducted. In GeoRisk, segments a 25 m long parts of streams and ditches, or 25 m long and 1 m wide parts of the shoreline of larger water bodies (rivers, ponds, lakes)

Static approach
Within GeoRisk the static approach means the calculation of PEC_{ini} values assuming lentic water bodies with a depth / width ratio of 0.3 and simultaneous application on all fields.
Mit dem statischen Ansatz ist im GeoRisk die Berechnung von PEC_{ini}-Werten unter der Annahme von stehenden Wasserkörpern mit einem Tiefe/Breite-Verhältnis von 0.3 und gleichzeitiger Applikation auf allen Flächen gemeint**.**

Trait, ecological trait (Schulz et al. 2007)
(Ecological) characteristicum of species, e.g. mobility, dispersal potential, Iuvenil development time, offspring number, generation time, survival rates, etc.
(Ökologische) Eigenschaft einer Art, z.B. Generationszeit, Wanderungsfähigkeit, Schlupfzeitpunkt, Überlebensrate, Nachkommenzahl usw.

Time Weighted Average (TWA)
Mean of a variable over a spefic time span where the values are weighted according to the time interval between them
Mittelwert einer Varaiblen über eine bestimmte Zeitdauer, wobei die Einzelwerte nach dem Zeitintervallen zwischen ihnen gewichtet werden

Time over Threshold (ToT or ToTh)
Time when the local PEC predicted by the dynamic approach is above an ecotoxicological threshold concentration, here the RAC
Zeit, in welcher die locale PEC über einem ökotoxikolog. Schwellenwert liegt, hier der RAC.

Uncertainty (Hart et al. 2006)
Imperfect knowledge concerning the present or future state of the system under consideration; a component of risk resulting from imperfect knowledge of the intensity of effect or of its spatial and temporal pattern of expression.
Unvollständiges Wissen in Bezug auf den aktuellen oder zukünfitgen Zustand eines Systems, eine Komponente des Risikos durch unvollständiges Wissen über Intensität der Effekte oder deren räumlichen und zeitliche Ausprägung

Waterside vegetation
drift mitigating vegetation betweein the water body and the crop area'
Aufragende Ufervegetation (Schulz et al. 2007**):** Unter aufragender Ufervegetation wird Vegetation verstanden, die als driftmindernde, vertikale Barriere für den Eintrag von PSM in Gewässer durch => Abdrift fungieren kann (=> Faktor).

Geodata based Probabilistic Risk Assessment of Plant Protection Products

(Georeferenzierte Probabilistische Risikobewertung von Pflanzenschutzmitteln)

GeoRisk

UBA Project code 3707 63 4001, AZ: 93 112-5 / 9

Appendix A

A FRAMEWORK DOCUMENT ON THE IMPLEMENTATION OF A GEODATA BASED PROBABILISTIC APPROACH TO ASSESS THE RISK OF PLANT PROTECTION PRODUCTS USED IN ORCHARDS, VINE YARDS OR HOPS CULTURE FOR AQUATIC POPULATIONS

Authors

Kubiak R, Hommen U, Bach M, Classen S, Golla B, Guerniche D, Klein M, Krumpe J, Preuss TG, Ratte HAT, Roß-Nickol M, Schäfers C, Strauss T, Toschki A, Trapp M

Project coordination

Prof. Dr. Roland Kubiak

Institute of Agroecology, RLP Agroscience, Neustadt a.d. Weinstraße

Dr. Udo Hommen

Fraunhofer IME, Schmallenberg

11.10.2011

Fraunhofer IME

JUSTUS-LIEBIG-UNIVERSITÄT GIESSEN

RLP AgroScience

JKI Julius Kühn-Institut

Institut für Landschaftsökologie

gaiac

Institut für Strategien und

Institut für Umweltforschung

RWTH AACHEN

Forschungsinstitut für Ökosystemanalyse

Content

1 Introduction

This framework document is a central product of the GeoRisk project, describing the GeoRisk approach and how it could be implemented in the risk assessment for the authorization of plant protection products in Germany. It corresponds to task Y of the project specification (*Erarbeitung eines strukturierten und inhaltlich konsistenten Entwurfs für einen Leitfaden (Rahmenkonzeptpapier, „Guidance Document zur Implementierung des georeferenzierten probabilistischen Ansatzes in die Risikobewertung von PSM (Teilbereich: Einträge in Gewässer über Abdrift und Verflüchtigung/Deposition*).“

The GeoRisk project and this document focus on the risk assessment for entries via spray drift and volatilization and deposition into water bodies from hops cultures, vineyards and orchards. However, the general approach should be applicable also to other entry routes and field crops.

This document summarises the scientific basis of the GeoRisk approach, i.e. model assumptions, data requirements, and hotspot criteria, and includes regulatory aspects, e.g. definition of the protection aim and protection level, hotspot management options and product specific risk mitigation options. A detailed description of the results of the GeoRisk Report can be found in the GeoRisk Main Project Report including its annexes (Kubiak & Hommen 2010).

The framework document developed by UBA, BVL, BBA and IVA (2006) was used as the starting point for the development of this framework document and refined based on the results of the GeoRisk project (Kubiak & Hommen 2010) as well as other related projects or publications.

1.1 Objectives of this framework document

The objectives of this framework document are:

- to summarise the final model assumptions and parameters to estimate local aquatic exposure resulting from spray drift entries from permanent crops,
- to summarize the evaluation of the uncertainty of the model inputs and implications for the protectivity of the risk assessment,
- to describe options for decision criteria for authorisation of plant protection products,
- to list options for local, not product related risk management (hotspot management) and product related risk mitigation measures,
- to describe the technical implementation and
- to outline the steps for the political implementation of the approach.

1.2 Background

Currently the aquatic risk assessment for authorisation of plant protection products in Germany is – similar to the approach in the EU – based on a realistic worst case scenario approach: The expected concentration of an active substance in edge of field water bodies (PEC = predicted environmental concentration) considers entries via drift, volatilization and deposition, runoff and drainage by the means of different exposure models based on conservative assumptions with respect to the environmental conditions. For example, to calculate drift entries it is assumed that during application the wind is always blowing from the treated field in direction of the nearest water body and that there is no drift mitigating vegetation between the crop and the water body, a static ditch of 1 m width and 30 cm depth.

Thus, the current approach is based on a worst-case scenario. However, in the meantime, the data and tools are available to make these assessment more realistic by the use of geodata to consider e.g. the real spatial relation between crop areas and surface waters and probabilistic approaches to consider quantitatively the variability of and uncertainty associated with the different parameters affecting the entry of plant protection products into water bodies and the exposure of aquatic populations.

Thus, the main aim of the GeoRisk approach is to use these data and approaches to achieve a more realistic spatially differentiated assessment and management of the aquatic risk. By refining the current worst case scenario in this way it is also aimed to allow reducing mitigation measures outside the identified high risk sections (‘hotspots’) while ensuring the protection of the aquatic populations. It is expected that such locally differentiated management will increase the acceptability of the mitigation measures by the various stakeholders.

1.3 Legal requirements and basic assumptions

The approach proposed here

- has to ensure that under the conditions of application no enduring negative effects for populations of non-target organisms occur,
- is based on the on the common prerequisite for ecotoxicological edge-of-the field assessments that usually populations rather than individuals are biological entity of the protection and that short-time effects can be acceptable (community recovery principle)[1],
- has to be in line with the German plant protection act,
- has to be in line with the EU Directive 2009/128/EC (EC 2009a) and EU Regulation 1107/2009 (EC 2009b), and
- has to consider the requirements listed in the framework document (UBA/BVL/BBA/IVA 2006, see 1.1): The information used for a decision on product authorization must be documented in

[1] However, for endangered species also the loss of individuals can be considered to be critical for the sustainability of the population (Liess et al. 2010) and also for vertebrates mortality of individuals might be unacceptable even if not relevant for the population.

such a way that it is clearly demonstrated that only scientifically sound, plausible and valid data have been used. In addition, all stakeholders, especially the applicants, must be able check the information, data and methods used in the authorization process.

2 Overview on the GeoRisk approach

Within the GeoRisk project a new approach for the aquatic risk assessment of plant protection products in Germany was further developed (based on Schulz et al. 2007) and evaluated. The aim was to establish a more realistic spatially explicit risk assessment. GeoRisk focuses on the risk assessment for drift entries from hops, orchard or vine culture and the approach is based on five key elements:

1. Geodata based probabilistic calculation of the potential entry of plant protection products into surface water bodies resulting from spray drift and volatilization.
2. Consideration of dispersion and transport of plant protection products in lotic waters (streams and ditches) by a dynamic model. For large rivers, lakes and ponds only the 1 m close to the shoreline is considered as a lentic water body.
3. Identification of ecologically critical aggregation of water body segments with high risks (hotspots), considering the recovery potential of affected populations.
4. Local, not substance specific risk management based on generic hotspot identification.
5. Authorization of products based on a substance-specific risk assessment with respect to the allocation of hotspots generated by product related hotspot identification.

The flow of the GeoRisk approach is outlined in Figure 1-1 and the following sections 2.1 – 2.5. A more detailed description is given in the sections 3 - 5.

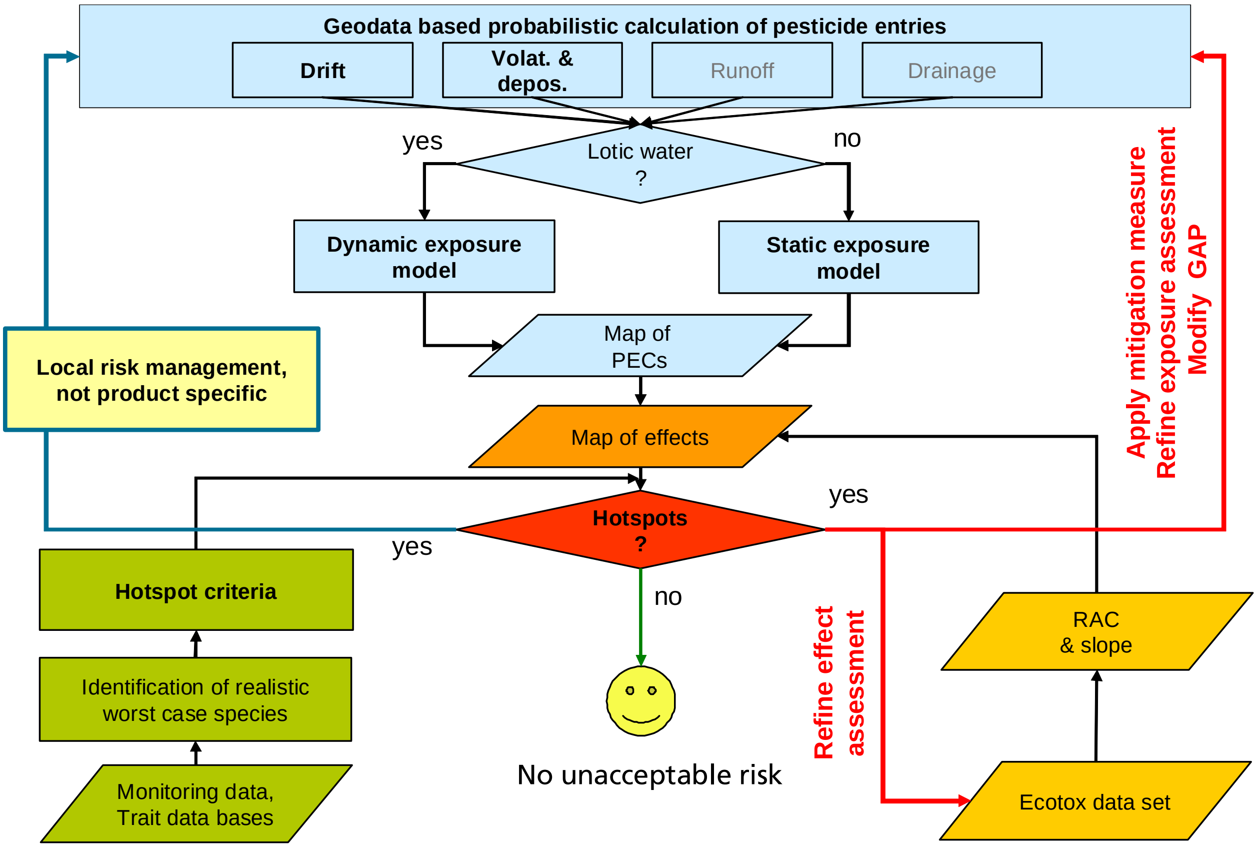

Figure 1-1: Overview on the GeoRisk approach for the aquatic risk assessment of plant protection products

2.1 Geodata based probabilistic calculation of pesticide entries

Water bodies included in the ATKIS dataset (streams, ditches, rivers, ponds, lakes, reservoirs) are divided into **segments of 25 m length as the basic units for the assessment**. For each of these segments (within a **distance of up to 150 m** to the agricultural area considered as relevant for potential drift entries) the drift load is calculated based on geodata in a partly probabilistic way. Spatially explicit data are given e.g. for the distance to crop in eight directions and the presence of hedges or other drift reducing structures. A probabilistic approach is used for the wind direction at the time of application and the drift deposition rate. For other parameters, e.g. the drift mitigation by a hedge, deterministic estimations are used. By means of first order Monte-Carlo simulations a distribution of the drift entries into each segment **(local drift entry distribution)** is calculated and a specific centile (i.e. the 90th centile) is used for the PEC calculation later on. The choice of the 90th centile is only determined by the regulatory requirement regarding the protection level which has to be achieved with the final risk assessment: The protection goal is to guarantee that unacceptable effects on the populations in the aquatic ecosystems due to the intended use of the evaluated product can be excluded with high level of certainty (i.e. 95%). It is assumed that by **use of the 90th centile of the local drift entry distribution** this goal can be achieved due to conservative assumptions in other parts of the assessment (see section 3.1 for details).

2.2 A dynamic exposure model for lotic waters

Only for the 1 m area close to the bank of larger rivers, lakes and ponds the currently used model of a lentic water body of 1 m width and 30 cm depth is used.

To consider that most water bodies in the agricultural landscape are flowing waters (streams and ditches[2]) with variable width/depth-ratios, a dynamic model was developed within GeoRisk to consider dispersion and transport. Additional geodata sets and algorithms were used to provide the necessary hydrological input parameters. Finally the model predicts exposure profiles over time for each water body segment depending on the stochastic drift entries up-stream. For the risk assessment these **exposure profiles are summarised into time weighted average concentrations (PEC_{TWA}) and times over threshold (ToT)**.

Due to the probabilistic nature of timing and magnitude of the drift entries in the upstream segments, Monte-Carlo simulations are necessary from which distributions of PEC_{TWA} and ToT for each segment are derived.

2.3 Hotspot analysis

By comparing the PEC calculated for each segment with the Regulatory Acceptable Concentration (RAC) segments of indicated risk can be identified (**Potential Risk Segment**, **PRS, characterised by PEC>RAC**). However exceedence of the RAC in segments can be acceptable if they are not expected to affect the sustainability of the aquatic populations (‘community recovery principle’). Therefore, GeoRisk has developed ‘hotspot criteria’ to identify those water body sections with an ecologically critical level of aggregation of risk segments. For **a generic, not product specific hotspot identification** the following hotspot criterion was derived:

Application of a product according to its intended use should not result in more than 10 % reduction of abundance of the populations in a 1000 m section of a water body.

For substance specific assessment, this criterion was further refined using a trait based approach to consider the potential of intrinsic recovery and recolonisation. Using monitoring data and trait data bases surrogate taxa were identified to represent realistic worst case combinations of traits relevant for the risk assessment. Based on the analysis of case studies of recovery and population modelling, **tolerable effect sizes were estimated for the surrogate taxa** (see section 3.5 for details).

In order to **calculate the expected effect** on a population on a 1000 m section, dose-response-relations have to be applied to the PEC-values of each segment. For generic hotspot identification

[2] The ATKIS object type ‘ditch’ is not restricted to linear lentic water bodies - ATKIS ditches can have considerable water flow.

a realistic worst case slope was suggested by the project while for product specific assessments the dose-response relations from the relevant ecotoxicological tests should be used (see 3.4.1).

In order to allow a more realistic but still protective assessment for the often very short exposure in flowing waters, an empirical model for **dependence of the RAC on the exposure duration** was developed (see 3.4.2).

2.4 Hotspot management

A generic hotspot analysis has to be conducted for all hops, orchard and vine areas to identify those water body segments where maximum available risk mitigation measures would be not sufficient to prevent unacceptable risks to the aquatic populations.

Local risk management to reduce the entries into these hotspots would then ensure an acceptable aquatic risk even if maximum available risk mitigation measures (here: reduction of previous maximum distance measure from 20 m to 10 m) were lowered.

This hotspot management should be conducted by local actors and includes verification of the local situation (e.g. verification of the local parameters used to calculate the drift entries), managing the hotspots and controlling the implementation of reduction measures.

2.5 Authorisation of plant protection products

As long as the hotspot management is not finalised, the current risk mitigation measure should be applied in the hotspot areas. Later on, after successful hotspot management, the reduced risk mitigation measures can be applied.

In contrast to the current approach based on the TER (Toxicity Exposure Ratio), the **decision on authorisation and necessary risk mitigation measures is based on the expected appearance of new hotspots by the intended use of the product**.

3 Description model assumptions and input parameters

3.1 Calculation of drift entries

The drift entry from a crop area into a water body depends on the type of the crop, the distance between crop and water body driving the deposition rate, the presence of any drift reducing structure between crop and water body and the use of risk reducing techniques. For the calculation of PEC-values for single water bodies within the landscape the input parameters can be divided into three groups:

- Geo-referenced data: distance crop area – centre of the water body segment in 8 directions from the segment, hydrodynamic parameters (width and depth of the water body, flow rate), presence of drift mitigating vegetation,
- Probabilistic: wind direction at time of application (simplified to 8 directions), deposition rate (variability of deposition depending on distance, based on drift trial data from Ganzelmeier et al. 1995, Rautmann et al. 2001)
- Point estimations: e.g. % mitigation by vegetation and application technique.

For each segment first order Monte-Carlo simulations are conducted to consider the variability of the wind direction at time of application and of the deposition rate as found in the BBA field trials. These simulations result in a local distribution of drift entries[3]. To ensure the overall protection level of 95 %, **the 90th centile of this distribution is used to characterise the potential drift entry into each segment**. Thus, in 9 of 10 application events, the entry is expected to be below this value. Due to other more conservative assumptions (e.g. drift mitigation by vegetation and drift application techniques) in the total approach the use of the 90th centile of the local entry distribution is considered to be sufficient to achieve the intended protection level. In addition, the effect assessment is based on the assumption that <u>every water body segment receives the 90th centile of its local drift entry distribution</u>. However, even if the drift entries of segments close to the same field are correlated, it is very unlikely that e.g. all segments receive high entries. Thus, the resulting spatial distribution of PEC values based in these local 90th centiles represents a conservative estimation of the PEC distribution. However, the degree of conservatism could not be quantified within the project.

The following table summarises the use of input parameters for calculating drift entries. The full description can be found in section 4.2 of the main GeoRisk report (Kubiak & Hommen 2010).

[3] In first order Monte-Carlo simulations variability and uncertainty are not differentiated. Therefore, no confidence bands around the resulting distribution or confidence intervals for specific centiles can be calculated.

Table 3-1: Comprehensive overview on parameters and variables used to calculate drift entries into a water body segment (for more details see chapter 4.2 of the main report)

Variable, parameter	Probabilistic/ deterministic[a]	Protectivity, percentile[b]	Geo-referenced[c]	Methodology, value(s)
Location of water bodies	d	no[d]	yes	ATKIS analysis; HR analysis
Segmentation	d		yes	Default length: = 25 m but end segments can be shorter
Wind direction	p	no[d]	no	Distance analysis for n = 8 directions
Wind speed	p	no[d]	no	Variability of wind speed is not considered explicitly but included in drift deposition trial data which cover a range of wind speeds)
Distance crop – water body y	d	yes[b]	yes	GIS analysis for 8 directions, reference point: centre of river segment; Protectivity: distance measured from edge of field to edge of bank (not edge of water surface)
Maximum considered distance for calculation of drift entries	d	yes[b]	no	150 m, protective because no relevant drift over more than 150 m
Deposition rate	p	yes[b]	yes	Distribution function of spray drift deposition recalculated by MC from original field trial data. Geo-referenced with respect to the variable "distance edge of field – water body" (function itself is fixed, not georeferenced)
Drift reducing sprayer technique	d	yes[b]	no	Fixed factors 75, 90% (minimum of the mitigation achieved by the technique)
Drift reduction by shielding waterside vegetation (hedges, windbreak)	d	yes[b]	yes	Fixed factor 25% for MS identification. Protectivity: reduction factor is higher during summer and autumn. For authorization purposes reduction during the year is expressed with a trapeze function
Drift reduction by emerse vegetation and shielding herbs	-	yes[b]	yes	Not considered yet Protectivity: smaller brooks with low flow velocity are often (at least partly) covered by emerse vegetation during summer and autumn
Deposition indicator for an individual water body segment	d	90[th]	no	Percentile the deposition distribution influence by "distance edge of field – water body", deposition rate associated to wind direction, stored in the data base protective because unlikely that every segment receives its local 90[th] centile For multiple application a lower percentile is used (according FOCUS, see chapter 6.4 of the report)

[a)] p: probabilistically distributed variable, d: deterministic variable;
[b)] degree of protectivity (conservatism) of parameter estimation: percentile of value (in case that an exact determination of a percentile is possible or defined by the methodology of derivation)
Protectivity "yes" means: the value(s) are chosen beyond the mean or median of its distribution, but their degree of probability (a centile) cannot be identified exactly.
[c)] Geo-referenced variable: do values of the variable differ for spatial units (regions, river branches, river segments)
[d)] As long as it is not satisfied that all water bodies are captured by GIS analysis without any exception it is to assume that among the undetected water bodies are some segments which are "at risk".

3.2 Entries via volatilisation

According to the model EVA (Holdt et al. 2010) deposition caused by volatilisation can be significantly higher than respected deposition caused by spray drift at least for volatile compounds.

Calculation of entries via evaporation can be conducted in the same way as described for the drift entries. The function for the deposition depending on distance is different, and also some other parameters might be different (e.g. mitigation by vegetation and technique).

A generic analysis was not conducted within the project: For the generic examination volatilisation cannot be considered as this process is highly substance specific. It is therefore recommended to consider this entry route on a substance base only. The process can be implemented rather easily as an additional load such as spray drift by selecting the crop- and substance specific deposition percentile together with the BBCH-stage at the time of application. Based on this information the deposition at 1 m can be calculated as a constant number. Dependent on the actual distance to the surface water body this amount has to be corrected within the geo-referenced system using the decline function of EVA (GeoRisk main report, chapter 4.3).

3.3 Calculation of water concentrations (PEC-values)

The calculation of the expected concentrations in a water body segment resulting from the pesticide entries are differentiated according to the type of the water body.

The water body close to the shore line of large water bodies like rivers and lakes as well as ponds is considered as a lentic (static) water body and the current standard water body model (1 m width x 30 cm depth with a rectangle profile, no flow) is assumed in order to protect the community in these ecosystem compartments for the shore line. One outcome of the current project was that these assumptions, compared with reality, are not always worst case.

For these water bodies initial water concentrations (PEC_{ini}) are calculated directly from the entry and the assumed geometry and used for the further risk assessment.

For other water bodies like streams and ditches[4] the hydrodynamic parameters (water depth, flow, etc.) are explicitly considered by a dynamic model. The characteristics of this new dynamic flow model are the use of near-to-reality flow rates and depths and a Monte-Carlo-based calculation to consider different possible application scenarios along the water body (Figure 3-1). Therefore it was assumed that all pesticide applications (of one product) along a water body are carried out during 2 days with two application windows of hour 1 – 10 and hour 24 – 34. In conclu-

[4] The ATKIS attribute „Graben“ (ditch) only indicates the function of the water body or that it has been artificially created or modified. It does not include information on the flow of the water. It was shown, that ditches according to ATKIS can have considerable flow (see GeoRisk main report). For example a water body directly adjacent to a roadway is defined as ditch even if the same water body in distance to the roadway is defined as stream.

sion the GeoRisk project proposes the implementation of a dynamic flow model. For three exemplary water bodies GeoRisk demonstrated that an implementation of such a dynamic model is in principle possible but further work is needed to make this method fully operational.

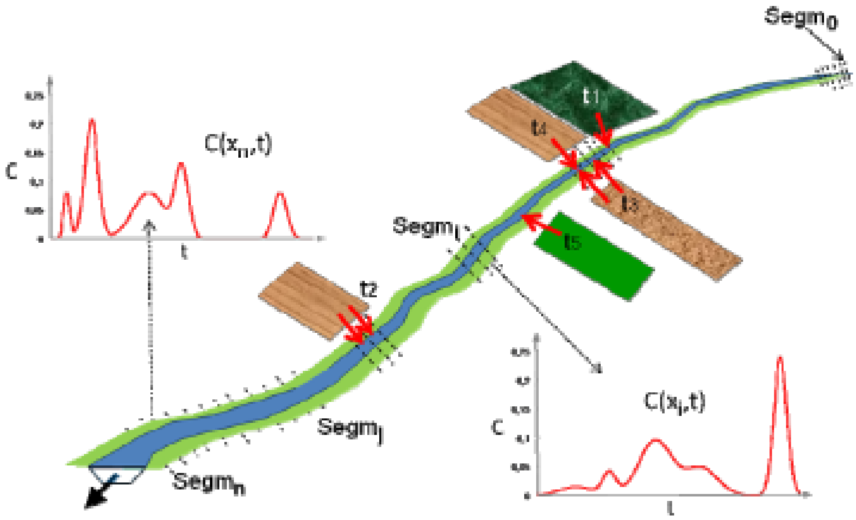

Figure 3-1: Conceptual scheme to consider pesticide deposition into a stream from different fields at different times t_n resulting in an exposure profiles $C(x_i,t)$ in each of the n water body segments

The assumptions used to calculate PEC-values for water body segments are summarised in Table 3-2. More details on the approach can be found in section 4.6 of the GeoRisk main report (Kubiak, Hommen et al. 2010).

Table 3-2: Comprehensive overview on parameters and variables used to calculate PEC-values from entry values (for details see chapter 4.4 and 4.6 of the main report))

Variable, parameter	Probabilistic/ deterministic[a]	Protectivity, percentile[b]	Geo-referenced[c]	Methodology, value(s)
Affected water bodies				
Type of water body	d	yes[b]	yes	a) **Lentic**: lakes (ATKIS object type 5112), rivers (5101) width of > 12m, type) b) **Lotic**: streams (5101), ditches (5103)
River network system	d		yes	a) static: consistently, (topologically correct) b) Dynamic: plus flow direction, discharge and closing of gaps (hydrologically correct)
Lentic water bodies (lakes, rivers, ponds): static exposure model				
Receiving water volume	d	yes[b]	yes	Only shore line considered:, no dilution 1 m width (affected part of a water body), 0.3 m depth, rectangle profile
Sediment concentration	---		---	sorption/desorption processes not considered (not relevant for short term exposure assess-ment)
Endpoint used for risk as-sessment	d	yes[b]	yes	PEC_{ini} = Deposition / Water volume for individual segment; protectivity results from ignorance of dilution within the larger water body and dissipation (adsorption to particles, photoly-sis, etc.)

Table 3-2 continued: Comprehensive overview on parameters and variables used to calculate PEC-values from entry values

Variable, parameter	Probabilistic/ deterministic	Protectivity, percentile[b]	Geo-referenced[c]	Methodology, value(s)
Lotic water bodies (streams, ditches): dynamic exposure model				
Receiving water volume	p	mean	yes	Function of hydraulic features of water body as listed below
Depth	d	mean	yes	Calculated acc. Manning-Strickler equation from geodata x,y,z
Width	d	mean	yes	Calculated acc. Manning-Strickler equation from parameters x, y, z.
Water body profile geometry	d	yes[b]	no	Geometric form of river bed: rectangular (more conservative with respect to max PEC_{TWA} than trapezoid)
Discharge	d	mean	yes	Primary data: catchment water balance and discharge model (precipitation data, GIS analysis)
Bed slope gradient	d	mean	yes	Primary data: DEM analysis
Manning's roughness coefficient	d	mean	no	Primary data: literature value (15 $m^{1/3}$ s^{-1})
Flow velocity	d	mean	yes	Calculated acc. Manning-Strickler equation
Dispersion coefficient	d	mean	yes	Calculated acc. Fischer et al. (1979)
Superposition of drift depositions	p	mean	yes	Binomial distribution: probability that a flowing water package receives n=0, 1, 2, … drift depositions
Treatment time frame for the application of all fields along river stretch	d		no	Currently 2 days are proposed with a 10 hour application time per day;
Endpoints used for risk assessment	p	yes[b]	yes	Max $PEC_{TWA(1h)}$ = $\Sigma(\text{Depos.})_L$ / $\Sigma(\text{water discharge})_L$ with L=length of water course of flow time 1 h ToT: Time over threshold = Total time of PEC above RAC
Effect of tributaries	--		--	Not considered yet: superposition of max $PEC_{TWA(1h)}$ at confluence and further downstream of two river stretches
Non-stationary hydrological conditions along river stretch	--		--	Not considered yet: river hydrology, especially max $PEC_{TWA(1h)}$ constant over river stretch

a) b) c) refer to Table 3-1

3.4 Calculation of effects per segment

The PEC-values (PEC_{ini} for the static or $PEC_{TWA(1h)}$ for the dynamic exposure model) calculated for each segment have to be transferred to expected effects on the aquatic populations using ecotoxicological data. In the following it is assumed that the RAC (Regulatory Acceptable Concentration) is extracted from the ecotoxicological dataset by applying safety factors considering the remaining uncertainties to the endpoints from the ecotoxicological tests. Thus, these safety factors correspond to the trigger values in the TER approach: The RAC corresponds to the TER multiplied by the PEC and divided by the trigger value.

The generic calculation of compounds was based on a virtual compound sprayed with an amount of 1000 g/ha which is still yet able to be registered under the current authorization scheme. For this generic assessment the RAC is assumed to be the lowest PEC which can be achieved by the current mitigation measures. For example, for an application rate of 1000 g/ha in hops the 90^{th} centile of the deposition for 20 m (the maximum mitigation buffer) results in a PEC_{ini} of 6 µg/L. Thus, this generic product could only be authorized if the RAC is >= 6 µg a.s./L (Technical risk mitigation can be ignored for generic hotspot identification).

Two aspects are considered to transfer PEC-values into effect values per segment: magnitude and duration of exposure.

3.4.1 Considering magnitude of exposure via dose-response relations

In order to calculate effect levels in a water body segment from the PEC, the following approach is suggested (see section 6.2.8.1 of the GeoRisk report (Kubiak, Hommen et al. 2010).) for more details):

- As a first conservative step, 100 % lethal effect for PEC > RAC can be used.
- If magnitude of exposure is indicated as multiple of the RAC it could easily be decided if a refinement of the slope would change the result (i.e. the number of management segments): If PEC-values are in most cases higher then ten times the RAC, the use of a dose-response function will usually <u>not</u> change the result significantly because 100 % effect can be assumed.
- If a considerable number of risk segments are characterised by PEC < 10 x RAC the analysis should be refined:
    - o To calculate the effect per segment from the PEC, it is suggested to use the **RAC as the EC_{10} of the dose response relation**: The RAC is assumed as 'safe' concentration, thus not leading to an unacceptable ecological effect. A 10 % effect is considered as an estimation of a potential effect at the RAC because NOECs and EC_{10} are often considered exchangeable in the risk assessment and 10 % mortality in the controls is often accepted for the validity of a test. In addition, a 10 % difference in abundance is usually not detectable in (semi-)field studies and thus thresholds derived from those studies are

based on larger effects. On the other side, a lower effect level assumed for the calculation of potential effects from a PEC and an RAC would be less conservative (dose-response curve is shifted to the right). Thus, the assumption of 10 % effects at the RAC was considered to be a reliable conservative estimation.

- For a **generic assessment** (hotspot identification) **a realistic worst case slope of a 2-parameter logistic dose response curve should be used**, e.g. 4 derived for carbaryl[5].
- For a **product specific assessment (registration)**, the **slope of the dose response curve for the most sensitive taxon should be used** if available and – as in the generic assessment - the RAC should be used as the EC_{10}. For example, in the standard acute risk assessment for fish, the RAC would be the lower of two LC_{50} divided by 100 and a dose-response function with an LC_{10} = RAC and the slope of the dose-response of the most sensitive fish should be protective for other fish species, too. The similar approach could be used for EC_{50} of invertebrates as well as algae and macrophytes (standard trigger 10).
- For long-term tests, e.g. the *Daphnia* reproduction test or fish juvenile growth test, the RAC is based on the NOEC divided by a factor of 10, but it could also be based on the EC_{10}. However, within the project the focus is on acute effects, respectively **also for an RAC derived from chronic studies, conservatively lethal effects are assumed in the hotspot analysis**.
- If the RAC is based on an SSD (Species Sensitivity Distribution) the approach would be the same (only the RAC is derived by the use of a smaller safety factor). With respect to the slope to be used, the steepest slope of the species tested in the SSD should be used for a worst case estimation.
- Micro- and mesocosm studies are often used to refine the risk assessment for algae, macrophytes and/or invertebrates. The NOECs for the different groups can be used in the same way as NOECs from a laboratory test but with a reduced safety factor (as derived in the usually risk assessment based on such a study). Thus, divided by the safety factor it could serve as the EC_{10} for the dose response curve for the hotspot identification. However, the NOEAEC of a micro- or mesocosm study is based on the recovery of the affected populations. Because recovery is included in the derivation of the hotspot criteria for the different groups, **the NOEAEC should <u>not</u> be used to derive a RAC for hotspot identification**. However, if the micro- or mesocosm study demonstrates faster recovery than considered in the hotspot criteria, refinement might be discussed on a case by case basis.

[5] If dose-response data for other realistic worst-case substances are available, this estimation of a realistic worst case slope could be refined.

3.4.2 Considering pulse exposure

The dynamic exposure model often predicts very short duration of relevant exposure (Time over Threshold = Total time of PEC above the RAC) compared to the exposure situation in standard toxicity tests. To consider the likely lower effects of shorter exposure the following suggestions for the consideration of exposure which is significantly shorter than in the tests conducted to derive the RAC are made based on a literature review of studies where effects depending on exposure duration were reported (for details see section 6.2.8 of the GeoRisk report (Kubiak, Hommen et al. 2010).):

- If the effect assessment is based on the **acute Daphnia test** over 48 h:
  - If the PEC is above the RAC for less than 24 h, the substance specific EC_{50} for 24 h should be used. Because uncertainty regarding latency of effects is not changed compared to the standard test, the trigger value of 100 should be applied.
  - If only the 48 h EC_{50} is available, a more realistic but still conservative estimation of a RAC* for short exposure (< 24 h) could be done by multiplying the standard EC50 by 1.6 (the median of the data analysed here) and keeping the standard trigger value. No results for effects on *Daphnia* after less than 24 h were available yet.
- If the effect assessment is based on **acute tests with invertebrates** over 96 h:
  - The RAC* for pulse exposure can be estimated by the following formula based on data for carbaryl shown to be protective for other substances:

$$\mathbf{RAC(t) = RAC_{96h} * 5.05\ t^{-0348}}$$

- If the effect assessment is based on acute tests with **fish** over 96 h:
  - For some substances no differences for 24 and 96 h – LC50 were found.
  - The factors between 1 h LC50 and 96h LC50s were determined to be:
    - 11.5 for organophosphates (fathead minnow and chlorpyrifos) and
    - 10.5 (malathion) – 128 in Guppy with daily observations only
    - 15 – 31 for pyrethroids (lowest factor: fathead minnow and fenvalerate)
    - 65 for endrin in fathead minnow.
- For **chronic endpoints**, e.g. NOEC for inhibition of reproduction, growth and development or population growth rate (algae, *Lemna*), the recommendations of the eLink workshop (Brock et al. 2009) should be followed, i.e. to decide if and for which time window the time weighted average (TWA) concentration should be used.
- If the risk assessment is driven by a NOEC from a **micro- or mesocosm study:**
  - First the RAC should be compared to the PEC_{ini} respectively the maximum $PEC_{TWA(1h)}$ h from the dynamic model.
  - If this indicates a risk and the predicted exposure is significantly shorter than measured in the study, the TWA approach could be used as for chronic tests, see the recommendations of the eLink workshop (Brock et al. 2009).

3.5 Hotspot criteria and hotspot identification

In order to protect populations, the predicted effects in the water body segments have to be considered on a relevant spatial scale for a population. Thus, a **hotspot is defined as a section (a number of connected water body segments) where the predicted effect on a population is above a tolerable level.**

A first hotspot criterion proposed by the UBA was analysed and refined within the project. This first generic criterion assumed a relevant spatial scale (length of the moving window[6]) of 1000 m and a tolerable effect level at this scale of 10 %.

Sensitivity analysis indicated that the total length of hotspots was not very sensitive for the spatial scale considered to be relevant for the population. Additionally it was found in the literature that even populations with low dispersal abilities are able to manage distances of 1 km. On the other side, a larger window size for the hotspot criteria will always reduce the number of management segments.

Therefore, 1 km is used as the relevant spatial scale for the hotspot criteria.

The refinement of the tolerable effect considering recovery of populations was based on a review of case studies on recovery, monitoring data to identify sensitive trait combinations and population modelling (for details see section 6.2 of the GeoRisk report (Kubiak, Hommen et al. 2010).).

The review revealed that for **phytoplankton (also periphyton) and zooplankton** after single reductions of abundances up to 90 % recovery within one year can be expected in almost all cases.

For **macroinvertebrates**, sensitive species representative in terms of their trait combinations were identified from monitoring data sets. Using a modelling approach taking the life-cycle of realistic worst-case species into account, it was demonstrated that effects on populations due to one yearly application can increase over time, which strongly indicates that the effects depend on the life-cycle characteristics. Strongest effects were found in species (long iteroparous) with a long life span and a juvenile development lasting more than 2 years, whereas for univoltine species (short semelparous) recovery within one year was demonstrated. Sensitivity analysis of the model indicates that calculated population effects as model output proved to be robust for the assumptions made for model calibration.

Additionally it was demonstrated that timing of application and spatial exposure patterns influence the overall effects on population in the field. For both of these factors worst case assumptions were used to calculate the tolerable effects on the population level. The results from this analysis indi-

[6] The term „moving window“ is used according to the UBA proposal (2007). It does not explicitly refer to a raster neighbourhood function but is used to describe the approach of analyzing connected water body segments according to given criteria.

cate that for the generic approach only 10% mortality for the population of the long iteroparous trait group is acceptable and should therefore be used.

For **macrophytes** and **fish**, the first generic criterion with a tolerable effect size of 10 % was kept because of the importance of macrophytes within the aquatic ecosystems (primary production, habitat, shelter, …) and the higher protection level and the higher protection level needed for vertebrates, if the focus is on the level of the individual rather than of the population (aesthetics).

Thus a generic hotspot criterion of no more than 10 % effect per 1000 m water body is proposed.

Only in product specific risk assessment where algae or zooplankton taxa are clearly the most sensitive taxa, a short-term effect of 90 % can be tolerated.

Otherwise also a 10 % tolerable effect criterion is recommended unless it can be demonstrated by higher tier studies that for the most sensitive taxa higher tolerable effect levels would not result in adverse effects on the population.

3.6 Consideration of multiple applications

For authorization of a specific plant protection product it might be necessary to consider **multiple applications**. Therefore it was calculated how large the effects of the single application can be to result in a given total effect under the assumption of independent effects of the single applications (no increased sensitivity of pre-exposed organisms but also no recovery of the populations between the applications). The following Table 3-3 provides the resulting **adapted tolerable effect levels to be used in the hotspot criterion** for the single application for different numbers of applications and different levels of total effects.

Table 3-3: Consideration of multiple effect
Centile of the deposition distribution to calculate the entry resulting from one application (according FOCUS to result in in total of the 90th centile for all applications and tolerable effect levels for the single application for different levels of total effects and different numbers of applications per year under the assumption of independent effects of the single applications.

number of applications	used centile of the deposition distribution	% tolerable effect								
1	90.0	10.0	20.0	30.0	40.0	50.0	60.0	70.0	80.0	90.0
2	82.0	5.1	10.6	16.3	22.5	29.3	36.8	45.2	55.3	68.4
3	77.0	3.5	7.2	11.2	15.7	20.6	26.3	33.1	41.5	53.6
4	74.0	2.6	5.4	8.5	12.0	15.9	20.5	26.0	33.1	43.8
5	72.0	2.1	4.4	6.9	9.7	12.9	16.7	21.4	27.5	36.9
6	70.0	1.7	3.7	5.8	8.2	10.9	14.2	18.2	23.5	31.9
7	69.0	1.5	3.1	5.0	7.0	9.4	12.3	15.8	20.5	28.0
8	67.0	1.3	2.8	4.4	6.2	8.3	10.8	14.0	18.2	25.0

Thus, if for example a pesticide should be applied three times per season, and the tolerable effect level for the relevant taxon would be 30 %, the effect of a single application should not exceed 11 % (calculated as $tol\_eff_{n\_appl} = 100 - (100 - tol\text{-}eff_{1\ appl})^{1/n\_appl}$).

When the static exposure model is used (for lentic systems) it should be considered that it is unlikely that each of the multiple applications results in entry, respectively PEC_{ini}, equal to or above the 90th centile of the local distribution. This is also considered in the current approach by using reduced centiles to calculate the PEC_{ini} from the sum of the entries of the single applications (FOCUS 2001, see column 2 in Table 3-3). In contrast to the FOCUS approach where the single event PEC_{ini} is multiplied with the number of applications for the calculation of the TER, here the effect of the multiple applications is considered on the effect side by the reduced tolerable effect thresholds listed in Table 3-3. Thus, the PEC_{ini} of a single event should be used.

The dynamic exposure model used for PEC calculations in lotic waters provides per se time series of the PEC for each water body segment. Thus, it is straight forward to model the whole application period and to extract the maximum $PEC_{TWA(1h)}$ for the further assessment using the adapted tolerable effect levels. In a first step, the total ToT should be used for the calculation of the RAC_{dyn}.

For a **refined assessment**, the dissipation of the substance between applications could be considered, too. On the effect side, possible carry over effects (the same dose might have higher effects with increasing number of applications) and recovery between applications could be included. As for the total tolerable effect levels, the tolerable effect levels for multiple applications could also be refined by a higher tier assessments.

Table 3-4: Comprehensive overview on parameters and variables used for the effect assessment and hotspot identification (for details see chapter 6 of the main report)

Variable, parameter	Probabilistic/ deterministic[a]	Protectivity, percentile[b]	Geo-referenced[c]	Methodology, value(s)
Relevant exposure characteristics	d	yes[b]	yes	Static water body: PEC_{ini} Dynamic water body: max $PEC_{TWA(1h)}$, ToT (Time over threshold) protective because dissipation not considered
RAC	d		no	Derived as in the current approach from ecotox tests / SSDs / mesocosm studies using safety factors (triggers) assumed to be protective
Effect per segment = % reduction of abundance	d	yes[b]	no	Logistic dose response function effect % = $100/(1+(PEC/EC50)^{-slope}$ EC_{10} = RAC slope = 4 for generic assessment slope from ecotox test for product risk assessment EC50 = f(slope, EC_{10})
RAC_{dyn}	d	yes[b]	yes	Estimation of lethal effects of short term exposure RAC_{dyn} = empirical functions of RAC and ToT protective due to use of data for worst case substances
Spatial scale for hotspot criterion considered to be conservative	d	yes	no	1000 m (length of moving window) protective because larger scale would result in less hotspots
Tolerable effect level for hotspot criterion	d	yes	no	Acceptable product related single reduction of abundance: 10 % for macroinvertebrates and fish 90 % for phyto and zooplankton Refinement by higher tier tools possible
Multiple applications	d		no	Tolerable effects levels for single application events based on number of applications and total yearly tolerable effect Protectivity not quantified, single events considered to be independent
Hotspot criterion	d		no	Total effect within 1000 m water body above tolerable effect level
Consideration of sublethal effects	d	yes[b]	no	Considered as lethal effects (reduction of abundance)
Effects of entries of other products	-	no	-	As in the current approach, not explicitly considered
Recolonisation (over larger distances, e.g. from river not directly connected and/or from tributaries)	-	yes[b]	-	<u>Not</u> regarded (not applicable for generic trait based approach) Protective because recolonisation would increase recovery

a), b), c) refer to Table 3-1

4 Hotspot management

The spatially explicit analysis of exposure and effects allows the identification of water body sections with high risks (hotspots) in the landscape. This offers the opportunity to conduct a **local risk management (not related to specific products)** which – if successfully performed –generally reduces the local risk due to drift entries and thus, allows reducing the product related risk mitigation measures.

The following options for a hotspot management were identified:

Table 4-1: Overview and assessment of management measures to reduce spray drift contamination at surface water segments with a high risk (modified from Schulz et al. (2007) and GeoRisk workshop, UBA, Dessau, Nov. 2009)

<table>
<tr><th>Management option</th><th>Effectiveness</th><th>Practicability</th><th>Costs</th><th>Acceptance</th><th>Controlling</th><th>Time frame</th><th>Responsibility</th></tr>
<tr><td rowspan="2">Clearing of rows of treated crops adjacent to water body</td><td>Very high</td><td>Easy</td><td>Costly</td><td>Very little</td><td>easy</td><td>Short</td><td>Farmers LCA[a]</td></tr>
<tr><td colspan="7">Costs mainly caused by loss of productive land (loss of revenue)</td></tr>
<tr><td rowspan="2">Planting of hedges (between treated fields and watercourse</td><td>High</td><td>Easy</td><td>costly</td><td>?</td><td>easy</td><td></td><td>Farmers LCA[a]</td></tr>
<tr><td colspan="7">Effectiveness depends on features of landscape elements: height and density of hedges
Implementation: short term; achievement of effectiveness: medium-term</td></tr>
<tr><td rowspan="2">Installation of drift protection fences or nets (temporarily)</td><td>High</td><td>Medium</td><td>Medium</td><td>?</td><td>Easy</td><td>Short term</td><td>Farmers</td></tr>
<tr><td colspan="7"></td></tr>
<tr><td rowspan="2">Planting of natural river bank strips</td><td>High</td><td>Medium</td><td>Costly</td><td>?</td><td>Easy</td><td>Short-/medium-term</td><td>Farmers LCA[a]</td></tr>
<tr><td colspan="7">Implementation: short term; achievement of effectiveness: medium-term</td></tr>
<tr><td rowspan="2">Modification/omission of watercourse maintenance</td><td>Medium</td><td>Easy</td><td>None</td><td>?</td><td>Easy</td><td>Short term</td><td></td></tr>
<tr><td colspan="7">Reduction of costs (saving)
Effectiveness depends on features of landscape elements: height and density of natural river bank vegetation
Implementation: short term; achievement of effectiveness: medium-term</td></tr>
<tr><td rowspan="2">Improvement of recolonisation (melioration of structural river quality)</td><td>?</td><td>difficult</td><td>Costly</td><td>Good</td><td>Easy</td><td>Long term</td><td>Water authority LCA[c]</td></tr>
<tr><td colspan="7">e.g. by creation of refugia / sources for recolonisation, improvement of structural diversity etc.
Controlling of implementation: easy; controlling of effectiveness: difficult</td></tr>
</table>

a) LCA: Land Consolidation Authorities: Local authorities responsible for agricultural advise, for land management and for local environmental protection (different authority structures in the different German states).

For the implementation of such a hotspot management the following **stakeholders** have to be involved in the whole process of hotspot identification and management:

– Registration / risk regulation: BVL, UBA
– Administration: agricultural extension service of the federal states, local advisors
– Industry: IVA
– Farmer associations: orchard farmers, vine-growers, hop-growers
– Science: research institutes

A hotspot management should cover the following requirements:

– All stakeholders should exactly know where the individual hotspots are.
– The hotspot management has to be performed locally, but nevertheless has to be organized and supervised on a national level to make the ongoing actions and efforts transparent to all stakeholders
– A co-operation of all stakeholders must be organised to enable the financing of the hotspot management.

Taking into account the requirements above, the most viable solution for the implementation of the hotspot management seems to be the following proposal simplified in Figure 4-1.

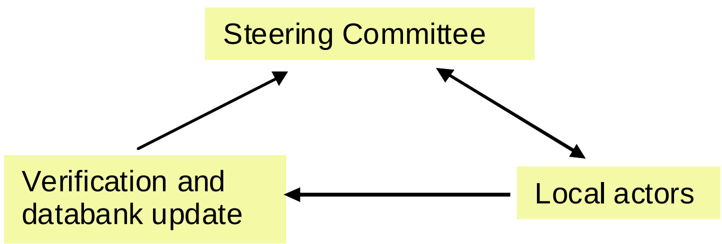

Figure 4-1: Proposal for implementation of hotspot management

The **steering committee** (SC) should be responsible for the overall organisation and the yearly budget for the hotspot management. The head of the SC should be a representative of the BVL. Members of the SC should be heads of state agricultural advisory boards, UBA, JKI, IVA and from academia.

Local actor groups will consist of agricultural advisors, members of local governments, local administration, farmers and scientific advisors. The local actor groups should find agreements with the SC about the way to handle the hotspots and to do the work.

Verification of hotspots in the field **and databank update** should be done by a scientific institute with expertise in GIS work. This institute would be responsible for hotspot identification, control of hotspot management and updates of the databank with the geodata used for the risk assessment after a hotspot is successfully managed.

The **work flow for managing a hotspot** should include:

- Scientific institute gives the information about a specific hotspot to the SC
- The respective head of the state agricultural advisory forms a local acting group with the help of the local agricultural advisor
- The local group verifies the hotspot and makes a proposal how to manage the hotspot(s) if necessary
- The SC accepts or modifies the proposal from the local group. A common agreement is made between SC and the local group.
- The SC gives the budget for the management measures and the work is done by the local actor group.
- After a successful end of the work, the local actor groups reports the results to the SC and the scientific institute responsible for verification and databank update.
- The institute controls the success, confirms the effort and updates the GIS databank
- This is reported to the SC and the SC closes the file.

Within the GeoRisk main report it was estimated that it needs about 1 Mio € / year to manage 200 km of hotspots during 5 years (see section 9.4). This could be a common financing with contributions from the federal state[7] and the plant protection industry. Contributions from the EU via cross compliance programs would have to be checked but seem to be possible under the current EU regulation (see also chapter 9.5 of the main report).

The authorization based on the new approach described here can be brought into force and be applied for all areas except the hotspots. **For the hotspot areas maximal buffer zones up to 20 m should be further valid until the hotspot management is successfully carried out**. After a 5 year period all hotspots should be managed and the new system is into force without exceptions.

The hotspot management itself should not be a part of the official authorization process. Nevertheless the authorization process can refer to the hotspot management measures and allows for a reduced application distance (up to 10 m) where no hotspot exists. That is equivalent to the “Biotopindex” (index to assess quantity and quality of different biotopes in a region), already into force in Germany. However, one major difference to the “Biotopindex” would be that the areas to increase this index can be created anywhere (e.g. where agricultural productivity is low) while the

[7] For example under the umbrella of the national action plan for sustainable use of plant protection products, ‘Nationaler Aktionsplan zur nachhaltigen Anwendung von Pflanzenschutzmitteln’, http://nap.jki.bund.de/.

hotspot management has to conducted at defined locations, i.e. where the pesticide entries are high.

5 Product authorization

For the product authorization described in the following it is assumed that the hotspots have been identified by a generic analysis (section 3.3) and successfully managed (section 4) to reduce pesticide entries in a way that no unacceptable effects are expected for a currently authorized worst case product.

5.1 Concepts and options for the risk assessment

In general two options are available for the product specific risk assessment to decide on authorisation and necessary risk mitigation:

A. Comparing a specific percentile of a PEC distribution over all (crop-relevant) water body segments with the RAC.

B. Check whether the intended use of the product would cause new hotspots in the landscape.

Option A was originally suggested during the UBA – IVA – BVL workshop (Klein et al. 2006) and was further developed in the framework document (UBA/BVL/BBA/IVA 2006), the former UBA-Project (Schulz et al. 2007) and also in a suggestion of the IVA during the GeoRisk workshop (Dechet 2009).

The general approach of option A is summarised in the following figure, details are given Schulz et al. (2009):

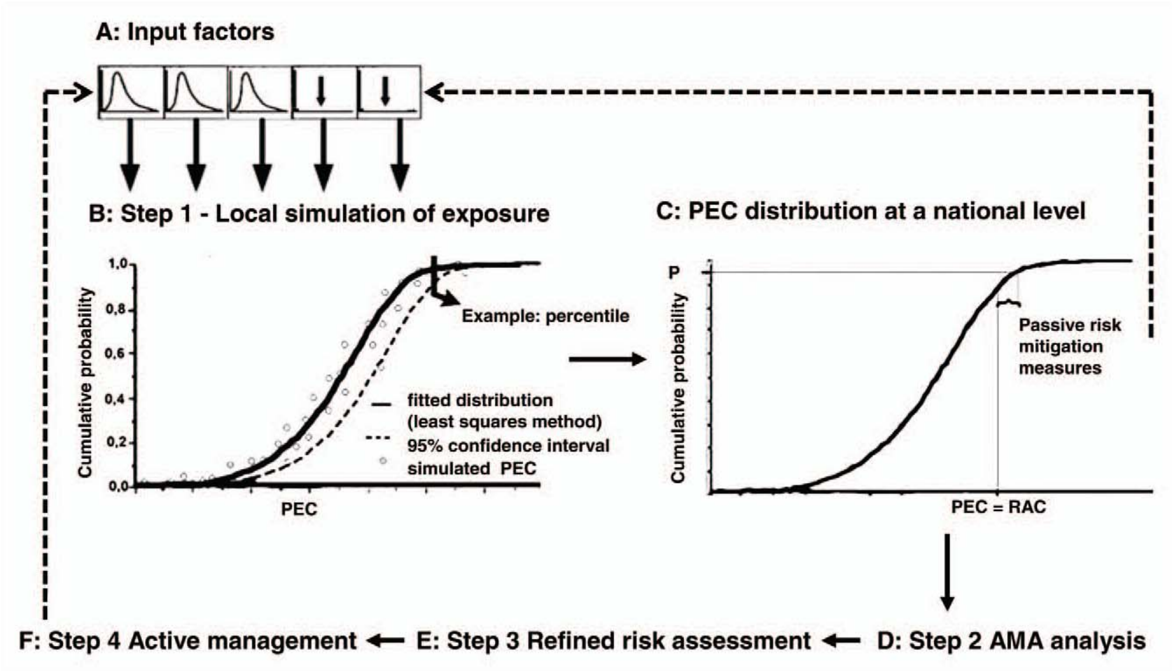

Figure 5-1: Scheme illustrating how and at which steps the setting of percentiles of exposure distributions occurs. The diagram also indicates that the number of active management area (AMA) management measures determined through a feedback step the setting of percentiles and confidence limits (from Schulz et al. 2009).

From the PEC distribution created for each water body segment (step 1 in Figure 5-1) the e.g. 90th centile is taken and combined to a new distribution, the PEC distribution at the national level. From this distribution again a specific centile is taken and compared to the RAC.

Some advantages and disadvantages of option A are listed in the following:

- Different risk mitigation options result in different landscape level PEC distributions and thus, similar to the drift tables used now, tables with landscape level PEC-values for the different combinations of crop, distance measure and centile can be created.
- These tabled PEC-values ('geoEckwerte_Aquatik', Dechet 2009) can then be used in the same way as the PEC values calculated in the current scenario based approach from the drift tables, e.g. by calculating a TER and comparing it to the specific trigger.
- The advantage of this approach is its simplicity for the user – the PEC values used for the risk assessment can easily be calculated from the tables and the application rate.
- However the disadvantage of this approach is that the spatial information is lost in the risk assessment and no link between the hotspots managed and the segments with PEC values above the RAC exists.
- In addition it seems not clear how the effects of a hotspot management should be considered in this approach. Successful hotspot management would affect the landscape level PEC distribution, i.e. the distribution will be narrower because exposure of the segments with the highest PEC values is reduced. Thus, the centile used for the comparison with the RAC will also be lower if the hotspot management is considered in the risk assessment. This would allow the

use of more toxic substances (lower RAC) but would also induce the need for a new hotspot management to protect these new segments with PECs above the RAC. Keeping the PEC-distribution based on the situation before the hotspot management seems not to be an option because the database must be regularly updated and refined.

– The approach is also difficult to use with the dynamic exposure model where exposure is characterised by magnitude (PEC_{TWA}) and duration (ToT). Consideration of the very short exposure in lotic waters is not possible because the relevant RAC_{dyn} is not a fixed value as the RAC but a spatially explicit parameter depending on the segment specific ToT.

Option B avoids the use of a landscape level PEC distribution by using the identification of hotspots due to the use of the product as the decision criterion. This would ensure that there are no product related unacceptable effects on local populations and an update of the geodata base (due to changes in landscape or availability of better geodata) does not affect the criterion.

However, testing the occurrence of new hotspots caused by the intended use of a product requires the calculation of the PEC values per segment, the transfer into effects and the application of hotspot criteria. Thus, the stepwise and spatially explicit risk approach of option B needs a more complicated tool as option A which is based on a simple table.

The tool must include a database of the expected generic PEC values for different product related risk mitigation measures and the option to calculate the product specific PECs from the application rate as well as the effects per segment based on the RAC, a slope of the dose-response curves and information to consider pulse exposure. In addition, the tool must allow for the hotspot identification (with the proposed criteria developed here but also with refined product, respectively taxon specific criteria).

In consideration of the regulatory needs of a geo-referenced risk assessment and the potential link to other legislative contexts in terms of local action plans, the GeoRisk consortium proposes option B, risk assessment based on the probability of new hotspots due to the use of a product, for product specific risk assessment of spray drift in vine, fruit and hop cultures in future.

5.2 Technical implementation

Within the GeoRisk project a web based tool was implemented containing a geodata base, the static exposure model, and an evaluation tool, e.g. for hotspot identification. The included database contains the necessary information for calculating drift entries per segment and also the resulting PEC_{ini} values assuming the standard water body properties used in the current scenario based approach (lentic water body with a width/depth ration of 3.3). The tool includes the routines that allow stakeholders to conduct an exposure and risk assessment (including application of hotspot criteria) and to download spatial data for further refinements using the model of lentic water bodies.

The workflow for building up spatial domain databases for the GeoRisk approach is described in the first part of the technical document (see Appendix B of Kubiak, Hommen et al. 2010). In the second part, the system documentation and user manual of GeoRisk-WEB is presented.

The creation of the necessary additional input data to run the dynamic exposure model for all lotic water bodies relevant for drift entries from permanent crops was out of scope of the GeoRisk project. Thus, this could be done in a follow-up project if the risk assessment should consider more realistic hydrodynamic parameters[8]. A mathematical model to generate a topologically correct flowing water network is developed but it has to be applied on all river basins in Germany relevant for risk assessment. For all river segments the set of hydraulic parameters (ref. Table 3-2) needed for the calculation of $PEC_{TWA(1h)}$ and ToT according to the dynamic approach have to be derived. This includes a substantial evaluation of the quality and uncertainty of all required parameters and geo-referenced variables, in combination with wide-spread ground-truthing and local measurements. Assumption on the frequency and the temporal distribution of substance applications have to be proofed by empirical data. Furthermore, an integral part of further studies will be a detailed mathematical handling of probabilistic $PEC_{TWA(1h)}$ and ToT calculation for dendritic river systems where substance concentration profiles from several river branches overlay. Finally the mathematical solution has to be operational to be integrated into a GIS database for practical application on large areas.

5.3 Product specific risk mitigation

A proposal for a new system of risk mitigation measures was out of scope of the project. However, the generic assessment and the hotspot identification were conducted under the assumed objective to replace the current 20 m distance measure by a maximum of 10 m. If one or two additional smaller distance measures (e.g. 3 and 5 m) should be introduced has to be decided separately as well as the classes of drift reducing techniques.

One proposal would be 3, 5 and 10 m distance measures which could be combined with 75 and 90 % risk mitigation by the specific application techniques.

[8] Within the GeoRisk project the necessary mathematical and hydrological concepts for the dynamic exposure model were developed and implemented as a prototype in ArcGIS . For a German-wide implementation of this dynamic modelling approach the necessary hydrological parameters have to be collected through ground truthing and to be combined with a GIS-based estimation using digital terrain models as well as information of third parties (e.g. federal and regional hydrological authorities).
In addition, geo-referenced information on distribution of ditches with more or less stagnant water is needed, but at the moment not available in the ATKIS DLM2-dataset (ditches according to ATKIS can have considerable water flow). Therefore this has also to be estimated or derived by ground truthing, or from federal and regional hydrological authorities.

6 Implementation of the GeoRisk approach for authorization of plant protection products in Germany

During the GeoRisk workshop with representatives of different stakeholder organisations (see Appendix C in Kubiak, Hommen et al. 2010) the following **roadmap to implement a geodata based probabilistic approach in Germany** was developed:

1. Joint elaboration of the parameters required for the determination of the hotspots based on the dynamic model for a more realistic simulation of lotic water bodies.
2. Calculation of the management segments for a pilot study area (e.g. the Hallertau) and, if possible, at least one further area with a different permanent crop.
3. Decision about the establishment of a pilot project (hotspot management and use of new mitigation measures) on this basis.
4. Performance of a pilot project, a field test e.g. in the Hallertau as a clearly defined area characterised by one specific culture, including a chemical and biological monitoring.
5. Decision about the implementation of the procedure starting with all orchard, wine and hops cultures.
6. In general there is a demand for an active coordination body for the implementation of a georeferenced probabilistic approach, e.g. by the BVL steering committee "Probabilistik".

7 Abbreviations

ATKIS	Amtliches Topographisch-Kartographisches Informationssystem
BBA	Biologische _Bundesanstalt
BVL	Bundesamt für Verbraucherschutz und Lebensmittelsicherheit
EC_x	Concentration resulting in x % effect
EU	European Union
GIS	Geographic Information System
$HC_{5^\wedge}$	Hazardous Concenration for 5 % of the species, 5^{th} centile of a SSD
HR	High resolution
IVA:	Industrieverband Agrar
JKI	Julius Kühn Institut
LCA	Land Consolidation Authority (Local authorities responsible for agricultural advise, for land management and for local environmental protection), different authority structures in the different German states
MC	Monte-Carlo simulation
NOEAEC	No Observerd Ecologically Adverse Effect Concentration
NOEC	No Observed Effect Concentration
PEC	Predicted Environmental Concentration
PEC_{ini}	PEC calculated from pesticide entry in the static exposure model related to the water volume
$PEC_{TWA(1h)}$	Maximum time weighted average PEC over one hour calculated by the dynamic exposure model
PRS	Potential Risk Segments (PEC > RAC)
RAC	Regulatory Acceptable Concentration (Toxicity value divided by a safety factor respectively trigger value
RAC_{dyn}	Dynamic RAC, considering reduced effects of pulse exposure
SC	Steering Committee
SSD	Species Sensitivity Distribution
TER	Toxicity Exposure Ratio
ToT	Time over threshold = Total time of PEC above RAC, calculated by the dynamic exposure model
TWA	Time Weighted Average
UBA:	Umweltbundesamt (Federal Environment Agency)

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GeoRisk

Geodata based Probabilistic Risk Assessment of Plant Protection Products

(Georeferenzierte Probabilistische Risikobewertung von Pflanzenschutzmitteln)

UBA Project code 3707 63 4001, AZ: 93 112-5 / 9

Appendix B

Technical documentation

Part I
Creating spatial domain data for geoRISK

Part II
geoRISK-WEB system documentation

Version: 0.9.3

Authors

Burkhard Golla, Jens Krumpe

November 30. 2010

Part 1 Creating spatial domain data for geoRISK

Part 2 geoRISK-WEB system documentation

1 Creating spatial domain data for geoRISK

1.1 Introduction

The first part of the technical guidance document serves two purposes: first, it describes the steps, that were taken to create spatial domain data for the geoRISK approach based on ATKIS BDLM data and the current ATKIS data model[1,2]; second, it enables UBA and other interested parties to build up such databases following a step-by-step procedure without being bound to specific GIS software products.

Figure 1-1 General workflow for creating the geoRISK databases

An overview of the general workflow is described in the final report, chapter 5.3. In the technical documentation the workflow is broken down to the single steps of the spatial data management process (Figure 1-2).

There are numerous commercial and open source software products that can be used for this task. (Steininger & Bocher 2009) give a detailed overview on existing free and open source desktop GIS projects. The GIS-Report (Harzer 2009) focuses on the description of commercial software solutions. WP2 used the functionalities of Oracle Spatial 11g for creating the spatial domain data.

1.2 Detailed workflow description

The detailed workflow description consists of ten topics. The process starts with the ATKIS BDLM data delivery from BKG and ends with the stream segmentation and the building of a topological network. In Figure 1-2 spatial feature data sets are symbolized by rectangles. Geometric processes are symbolized by rounded corner rectangles.

[1] Based on initiatives of the Federal surveying authorities the ATKIS data model is currently being replaced by the AFIS-ALKIS-ATKIS-data model (AAA-data model).

[2] Spatial data according to the new AAA-data model were not available at the beginning of the project and are today only available for selected states.

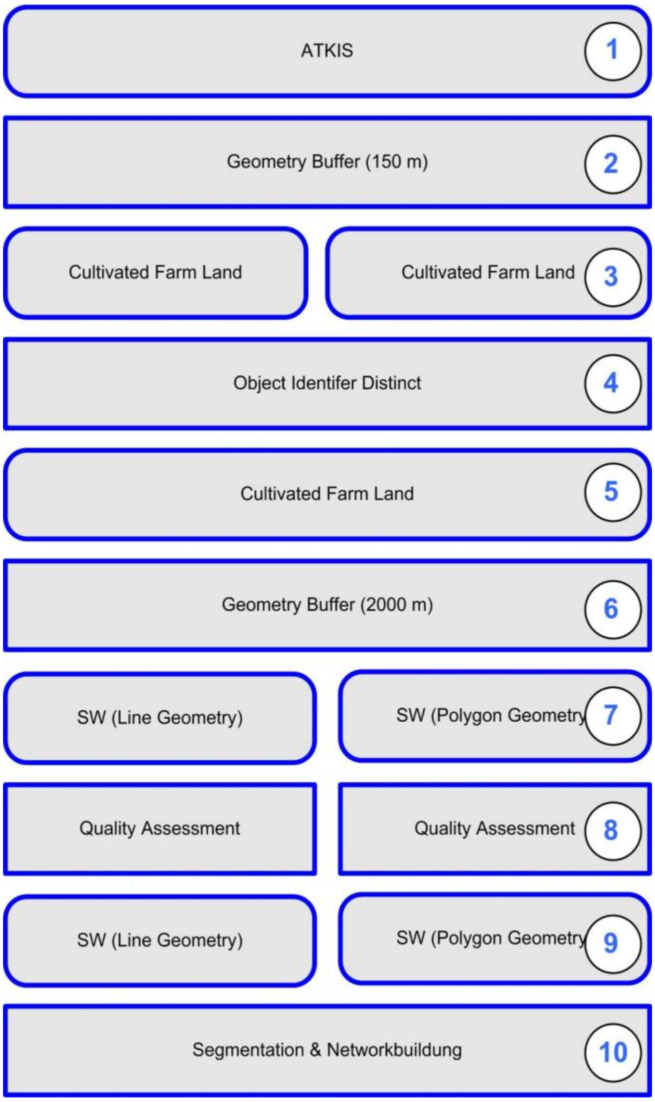


Figure 1-2 Steps of the spatial data management process.

1.1.1 ATKIS BDLM data import and validation process

The BDLM[3] data was provided by the BKG via UBA. The dataset for Germany is tiled into 9065 shape files. BDLM objects are grouped according to their thematic and geometry (point, line, polygon) in a layer structure (BKG 2005). The spatial extension of the tiles is a representation of the national TK100 sheet line system. The authoritative updating process of BDLM data mainly results from interpretation of aerial images. This periodical process defines the basic time reference of the data for a specific region (Figure 1-3).

[3] Provided by Geodatenzentrum [URL http://www.geodatenzentrum.de/ (requested on 28.2.2010)] via UBA

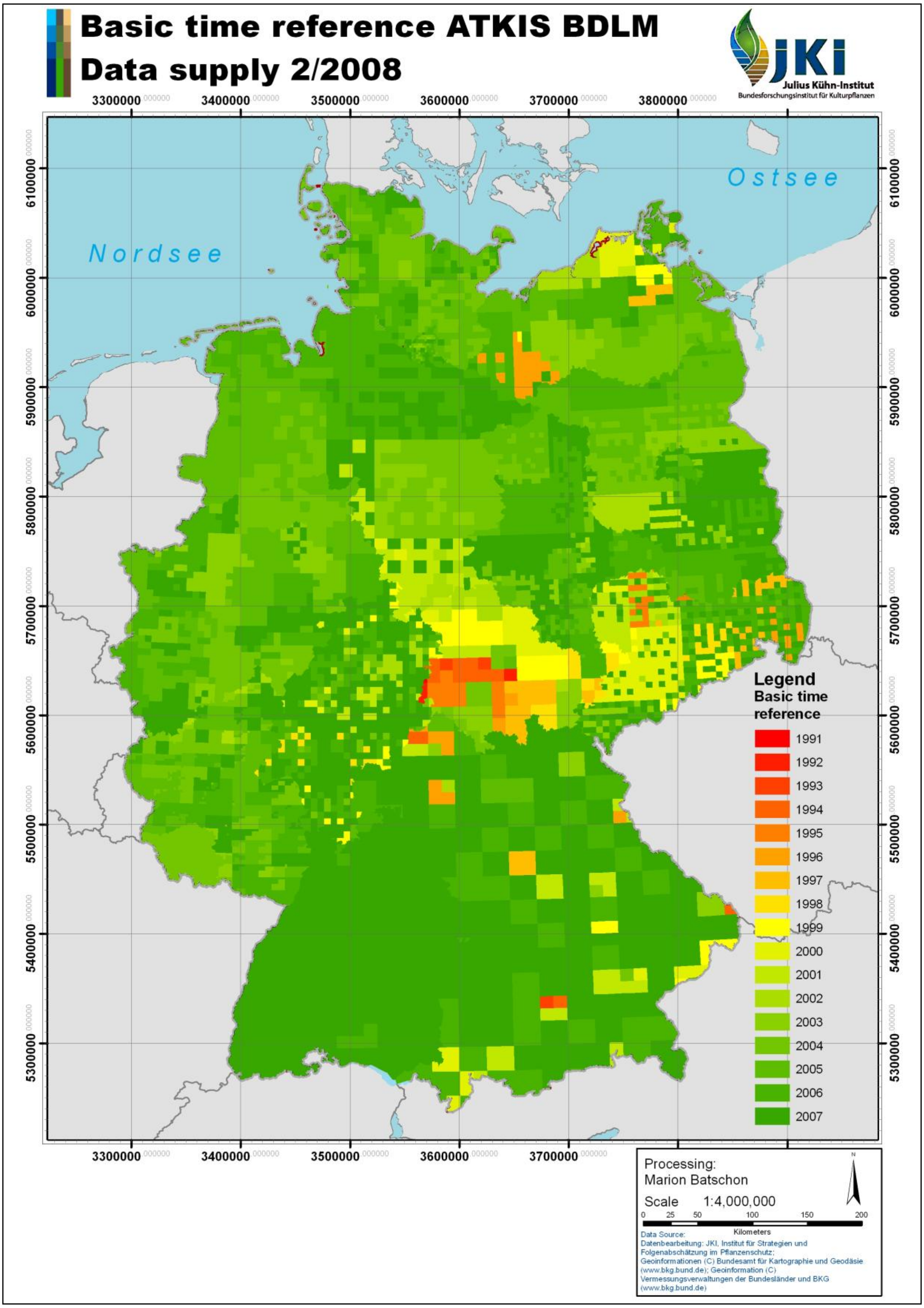


Figure 1-3 Basic time reference of ATKIS BDLM data delivery used in the project

After a tile-wise import to the spatial DB an automated consistency check of the imported features was performed for all ATKIS layers. The number of features in a layer in the database had to match the sum of features for this layer of all input files. In a second step a tile wise manual on-screen checks for visible white spots was performed to detect missing data.
On database side tiles with partly missing data were discovered which turned out also to be missing in the original shape files. UBA and BKG were informed. BKG delivered an additional set of DVD. The second data delivery in September 2008 was imported and checked with the same procedure. No errors were observed, but in later analysis it turned out, that especially in the Bavaria, overlapping of identical features occur. According to the information of BKG the reason for this is error lies in the process of taking over the data from the Bavarian Surveying Agency. These errors will not be fixed by BKG but will be eliminated with the fore coming AAA data model. For further data preparation and analysis within WP2 this circumstance did not matter, but this issue needs to be considered when performing e.g. area statistics.

Table 1-1 Area overlap in ATKIS_veg04 specialized cultivation (AOA4109)

ATKIS Object Type 4109	Original ATKIS BDLM data	(after dissolving overlaps)
KLT 2000 (HOPS)	ca. 355km²	ca. 195km²
KLT 3000 (VINE)	ca. 1.268km²	ca. 1.199km²
KLT 4000 (FRUIT)	ca. 92.728km²	ca. 89.119km²

1.1.2 Process of identifying permanent crop land in distance to surface waters

The new feature data sets are created by buffering BDLM line and polygon features (gew01_l and gew01_f) of surface water bodies separately (AOA 5101, 5103, 5112). The buffer distance for permanent crop land is 150 [m] and is measured in ground units. It was chosen an interpolation angle of 90 degree. All permanent crops which geometrically intersect the buffer features are selected.

1.1.3 Distinct feature data sets on permanent crop land in theoretical spray drift deposition zone

Two new area feature data sets are created representing crop land in a so-called theoretical spray drift deposition zone (Enzian & Golla 2006), for crop land resulting from the buffer around line and polygon features respectively. Within this zone there is a potential treat that spray drift exposure from crop land to surface waters might pose a risk to aquatic organisms. The definition stems from the context of a landscape percentile approach (ditto). Nevertheless this approach is used in the project for the identifying permanent cropland to be considered in the analysis.

1.1.4 Process of merging feature data sets and removing duplicates

Goal of this process is to create one feature data set on permanent crop land in theoretical spray drift deposition zone out of the two sets that result from step 3. Using distinct functionalities allows removing duplicates of crop land features represented in both input sets.

1.1.5 Single feature data sets on permanent crop land in theoretical spray drift deposition zone

One feature data set of unique crop land features the theoretical spray drift deposition zone around surface waters is created.

1.1.6 Process of identifying surface waters to be included in the hot spot analysis

In this process another distance analysis is performed. For the hot spot analysis (see chap. 3) surface waters included that are connected to those potentially being exposed by surrounding crop land. The new feature data sets are created by buffering the unique set of single feature data sets on permanent crop land in theoretical spray drift deposition zone. The buffer distance is 2000 [m] and is measured in ground units. An interpolation angle of 90 degree was chosen. All permanent crops which geometrically intersect the buffer features are selected. The intersect operation is performed for line and polygon features (gew01_l and gew01_f) of BDLM surface water bodies separately (AOA 5101, 5103, 5112).

1.1.7 Distinct feature data sets on surface waters to be included in the hot spot analysis

Two feature data set on surface waters are created that included line features and polygon features of surface waters respectively to be included in the hot spot analysis.

1.1.8 Process of quality assessment

As the features includes in the BDLM are not compliant to a simple feature concept a quality assessment was performed. The process includes the detection of geometric self intersections, line joining and disaggregation. These issues will be solves with the new AAA-data model.

1. Detection of geometric self intersection

Any linear feature that self-intersects is detected and automatically split into separate features, one per non-intersecting part. Each resulting feature has the total number of parts created from the original feature added as an attribute. Any area features that self-intersect create more than one area. These areas are gathered into an aggregate.

2. Line Joining and disaggregation

In the sub-process of line joining non-intersecting features are connected to larger features by remove insignificant nodes. Any nodes with only two lines connecting to them (“pseudo nodes”) are removed. Lines features remain broken at points where three or more lines features converge. Only features with have the same object properties fulfill the join condition. In the process of disaggregation aggregate feature are decomposed into their components.

1.1.9 Qualified feature data sets on surface waters to be included in the hot spot analysis

Two feature data set on surface waters are created as a result of the quality assessment. The data sets included simple line features and simple polygon features of surface waters respectively to be included in the hot spot analysis. The following tables show the results of assessment.

Table 1-2 Result of the quality assessment for line features

	HOPS	FRUIT	VINE
length [km] before QM	5.654	75.442	18.142
length [km] after QM	3.980	63.995	15.941
Difference	1.674	11.447	2.201

Table 1-3 Result of the quality assessment for polygon features

	HOPS	FRUIT	VINE
length [km] before QM	2.952	28.519	8.058
length [km] after QM	1.614	19.036	5.128
Difference	1.338	9.483	2.930

1.1.10 Process of segmentation and network creation

The qualified spatial data sets on surface waters to be included in the hot spot analysis are segmented in 25m reaches. For polygon features common boundaries were identified and removed by creating larger areas.

1.3 Data model GeoRISK

The domain database *GeoRISK* consists originally of five nested database schemas. A schema is a collection of database objects like tables, views and indexes and describes a logical group of tables for a specific theme. Due to the modifications concerning the target DMBS[4] Georisk schemas and data dictionary were simplified into one single schema which is and documented in Figure 1-4.

[4] It was decided by UBA to implement the application on a Oracle 11g without spatial option

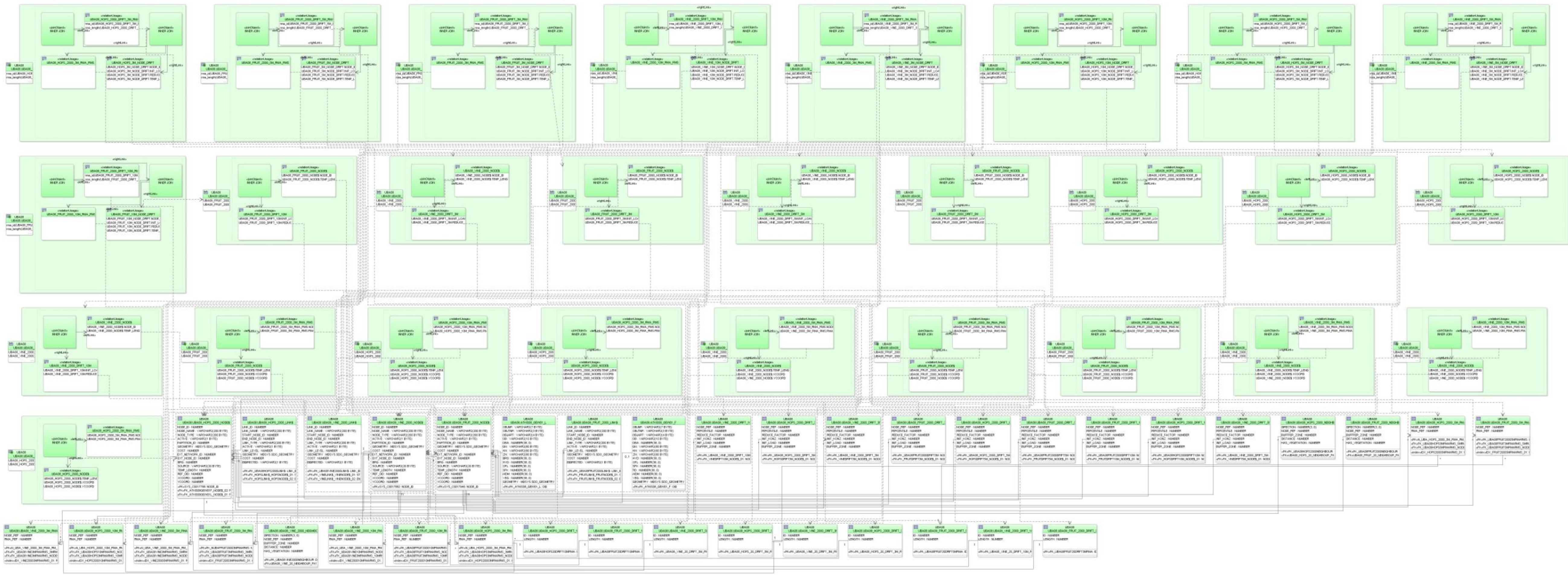


Figure 1-4 Georisk schemas and data dictionary for non-spatial Oracle DBMS

Table 1-4 Spatial geoRISK database schema (without system tables)

Schema Name	Description
LDB	The topographic_data schema stores all spatial data describing land use and land cover. The current project-status stores approximately 40GB vector data (ATKIS BasisDLM). The amount of active storage in this schema can increase. This depends on the selected index algorithm the final application. The topographic_data schema will be used for the spatial relation between water segments and adjacent landscape structure (filtering vegetation, application field etc).
NDB	The NDB schema is provided for future calculation of risk-management-segments. Wherever it used for network analysis, this schema contains all water segments. The current project-status stores approximately 1GB vector data. The amount of active storage in this schema can increase. This depends on which index algorithmic is selected for the final application. The network schema is designed to include networks for hops, grape-vine and fruit-crops.
EDB	The EDB schema represents the overall results of spatial exposure assessment. The EDB schema includes three different table-groups each for hops, grape-vine and fruit-crops.
meta_data	The meta_data schema stores further information about all data sources particularly spatial related data sources. If any table in topographic_data or network schema is manipulated an update of meta_data tables must occur.
authentification	The authentification schema stores the different user roles and groups in the database. This information will be used to separate access-functionality for the final web-based-application.

Table 1-5 Detail view on the EDB spatial schema using the example of hops (without system tables)

TableName	Description
NEIGHBOUR_HOPS	The neighbour_hops table describes the distance of water segments to the adjacent application area (hops). This table is linked in a one to many relations (compass-direction) to the node table, listed in network schema.
DRIFT_HOPS	The drift_hops table describes the 90^{th} percentiles which are calculated according to the geoRISK drift deposition module (see chaap. 4). This table stores the results using the real-distance. This structure takes into account to the requirements of integration high-resolution spatial data.
VIRTUAL_DRIFT_HOPS	The virtual_drift_hops table describes the 90^{th} percentiles which are calculated according to the geoRISK drift deposition module (see chaap. 4). This table stores the results of the probabilistic drift exposure for different buffer-zone-restrictions.

The network-components can in future developments be used for spatial network analyses. Spatial network databases are special cases of spatial databases. Their main feature is the combination of topologies and information about spatial locations (George & Shekhar 2007) The storage of the hydrological network-components are software independent with regard to the system used for network analyses. Therefore only standardized data types were used throughout the logical and physical database design. The spatial data type follows the guidelines of the Simple Feature Specification (OGC 1999).

It is recommended to update the network-schema separately from the data which belongs to the risk-assessment domain as there is no link between a geometric network representing small stream and spatial data of land use and land cover.

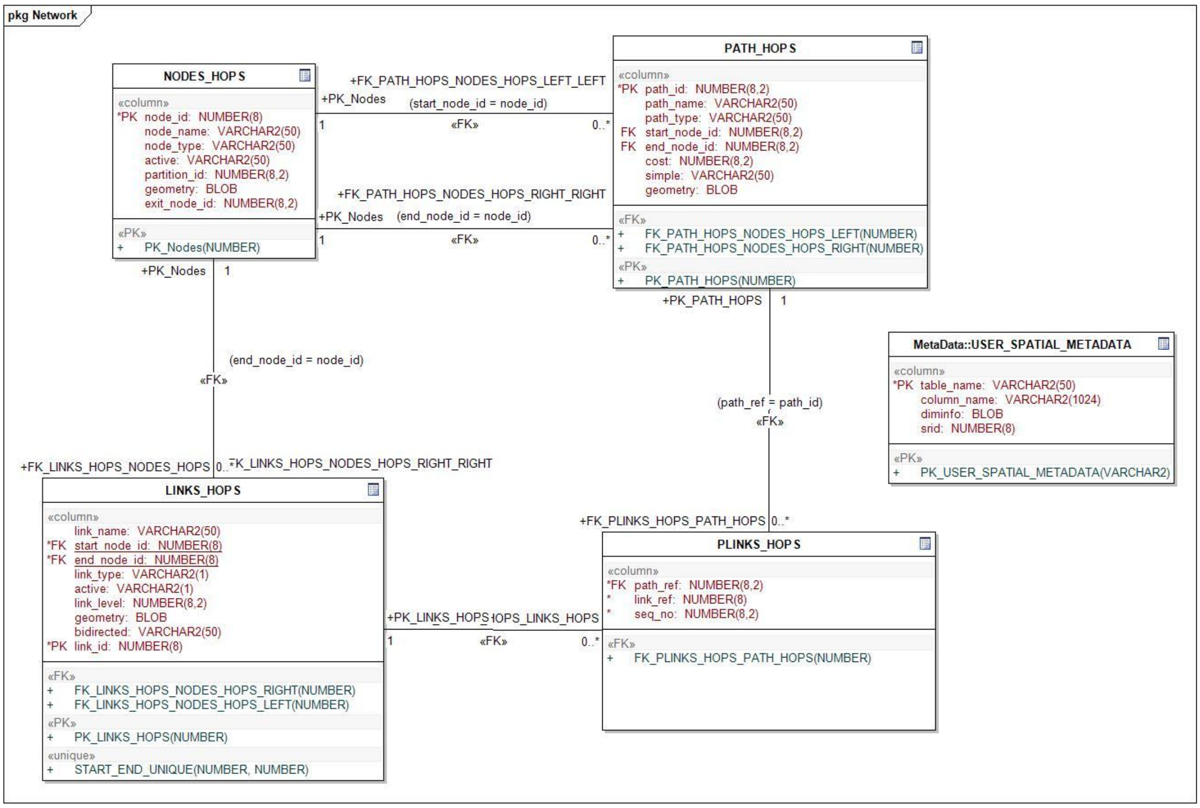


Figure 1-5 E/R diagram on the spatial network schema using hops as example (without system tables)

Table 1-6 Detail view on the spatial network schema using hops as example (without system tables)

TableName	Description
NODES	The node table describes all nodes in the network. One node corresponds to a water-segment which maximum length of 25 [m].
LINKS	The links table describes all links in the network. By definition, a link connects exactly two nodes.
PATH	The path table stores the start and end node of a path and its total cost. A

	path can be used for spatial indication of a risk management section.
PLINKS	The plinks table stores the list of all links that define a path.

The update of the network-schema mainly focuses of geometric correction of water segments. After any DML operation (update) is occurred, a new calculation of pesticide exposure in the risk-assessment schema must be triggered.

1.3.1 Back-up and historisation

For both database operations Oracle provides tools such as flashback data archive or Work space mananger for data historisation. Concerning compliance and efficiency the flashback data archive of Oracle 11g is suitable. It will not impact existing runtime performance. A new process - fbda (flashback data archiver) reads undo tables already generate and find undo for the tables of interest and stores the pre-image in a separate archive table. In contrast the work space mananger adds a trigger to turn updates into a flag delete + insert, a delete into a flag delete and modify the inserted data. This largely affects existing runtime performance especially for bulk operations. Additionally, the versions of the rows are kept in the same table as the current data with workspace manager, which potentially affect runtime query performance. With flashback data archive current transactions and queries will not be affected (Oracle 2010).

Part II

geoRISK-WEB

System documentation

Version: 0.9.2
(in German language)

1 geoRISK-WEB system documentation / System Dokumentation

Nach Pflanzenschutzgesetz {§§ 15 ff. PflSchG, 2004) ist das Umweltbundesamt (UBA) zuständig für die Sicherstellung des Schutzes des Naturhaushaltes vor unvertretbaren Auswirkungen bei oder als Folge der Anwendung von Pflanzenschutzmitteln. Hierbei kommt ihm die Aufgabe zu, sachgerechte Bedingungen (Auflagen und Anwendungsbestimmungen) für die Anwendung eines Pflanzenschutzmittels festzulegen. Die erforderliche Risikobewertung muss gemäß den gesetzlichen Vorgaben dem Stand der wissenschaftlichen Erkenntnisse und der Technik entsprechen.

Die Web-basierte Fachanwendung *geoRISK-WEB* versetzt das UBA und die am Zulassungserfahren beteiligten Institutionen in die Lage für konkrete Pflanzenschutzmittel Berechnungen durchzuführen, die nach dem Prinzip der „Managementsegmente (MS)" (chap. 3) in die Einstufung des PSM in eine Risikominderungsgruppen münden. Es können Berichte der Berechnungen erstellt und als pdf-Datei gespeichert werden. Entstehen im Ergebnis der Berechnungen potentielle MS, können diese als GML exportiert und außerhalb des Systems bspw. durch ein Geografisches Informationssystem weiterverarbeitet werden.

2 Program characteristics / Programmkenndaten

2.1 Identification of the Programm / Programmidentifizierung

2.1.1 Name of the program / Programmname

Der Titel der Anwendung ist geoRISK-WEB. Für die Applikationsentwicklung wird in Pfadstrukturen der Programmname uba08web verwendet. Die Quelltexte und Quelltextdokumentation sind auf dem beigefügten Datenträger in dem Ordner „/src“ enthalten.

2.2 Description of the program / Programmbeschreibung

2.2.1 Task of the program / Programmaufgabe

Das Programm ermöglicht es, anhand der Eingabe bzw. Auswahl von PSM-Merkmalen (Aufwandmenge (Application Rate (AR)), Regulatorisch Akzeptierte Konzentration (RAC) , Indikation (Wein, Obst, Hopfen), Anwendungsdatum) Gewässermanagementsegmente, in denen auf Grundlage der zugrunde liegenden Datenbasis, des Expositionsmodells (AP1, Standard Graben, chap. 4) und der PSM Merkmale rechnerisch die RAC in einer räumlichen Häufung überschritten wird („Hot-Spot Criteria“, chap. 3) zu identifizieren. Die Länge MS wird für die Risikominderungsgruppen (Risk Reduction Groups, RRG) in [KM] und in Ampelfarben dargestellt. Ein PSM kann nach dem Konzept des F&E Vorhabens bei 0 MS (Ampelfarbe grün) in der entsprechenden RRG zugelassen werden.

Außerdem ist in dem System eine Simulationskomponente enthalten, mittels derer sich für einen einzelnen Gewässerpunkt anhand dessen relativer Lage zu angrenzenden Applikationsflächen die Driftdeposition abschätzen lässt. Dieses Werkzeug steht nicht im Zusammenhang mit der Zulassungsprüfung sondern dient der Information. Die Simulation der Exposition kann für beliebig konfigurierbare räumliche Applikationssituationen vorgenommen werden. Aus technischen Gründen ist die Anzahl der Simulationen in der Anwendung auf max. 10 000 Läufe beschränkt, was nicht zu stabilen Depositionswerten in der zweiten Nachkommastelle führt.

2.2.2 Content of the programm / Programminhalt

Das Programm liest die Eingangsdaten AR, RAC, Scenario (Indikation), Date (Anwendungsdatum) ein. Anhand dieser Angaben erfolgt eine Datenbankabfrage, die für alle RRG das Ergebnis der MS Berechnung liefert. Diese Ergebnisse können 0 (potentiell keine Gefährdung), eine natürliche Zahl (Summe der potentiell gefährdeten Segmente in km) oder "out of range" für PSM, die eine zuvor festgelegte maximale Toxizität überschreiten, sein.

Die Simulationskomponente arbeitet mit einer Monte-Carlo-Simulation. In diese fließen die Abstände in acht Himmelsrichtungen zur nächsten Applikationsfläche, driftmindernde Vegetationen und der Tag der Applikation im Jahr ein. Ausgegeben werden die erwartete PSM-Beladung und -Konzentration des angegebenen Gewässerpunktes für frei wählbare Perzentile.

2.2.3 Special issues / Besonderheiten

Die Berechnungsdauer ist stark abhängig von den Eingangsparametern und vor allem von der Leistungsfähigkeit des Datenbankservers, sollte aber auf einem durchschnittlichen Server nicht die Zeit von einigen Sekunden bis wenigen (max. 15) Minuten überschreiten.

2.2.4 Size of the programm / Programmgröße

Die Größe des Programmes inklusive sämtlicher Bibliotheken beträgt etwa 30MB. Hinzu kommt ein Speicherbedarf für die Datenbanken von etwa 40GB.

2.3 Requirements of the program / Programmbedarf

2.4 Operating system / Betriebssystem

Das Programm funktioniert betriebssystemunabhängig.

2.4.1 Programming language / Programmiersprache

Das Programm wurde für Java 1.6 entwickelt.

2.5 Data organisation / Datenorganisation

Die berechnungsrelevanten Daten, sowie dazugehörige Metadaten werden vom Benutzer eingegeben oder gegebenenfalls aus einer Schadstoffdatenbank ausgelesen. Die Metadaten dienen als zusätzliche Information zu den Berechnungen und sind für die diese selber ohne Bedeutung.

Allerdings können die Metadaten bei einer Weiterentwicklung des Systems in die Berechnung mit einfließen und bilden so einen Anknüpfungspunkt für zukünftige Entwicklungen.

Anhand der Eingabe werden für die einzelnen RRGs die Ergebnisse aus einer Datenbank ermittelt. Diese Berechnungsergebnisse werden von dem Programm am Bildschirm ausgegeben. Zusätzlich kann der Benutzer einen Report im PDF-Format herunterladen, der sämtliche Eingabedaten und die Ergebnisse enthält.

Des Weiteren lassen sich die genauen Positionen der MS im GML-Format exportieren, um deren genaue Lage in einer GIS-Software darzustellen.

2.6 Responsibilities / Zuständigkeiten

Für eine robuste Internetanbindung, die Datensicherung, Server- und Datenbank-Administration ist der Betreiber der Anwendung/des Systems zuständig.

3 Functions of the program / Programmfunktion

3.1 Tasks / Aufgaben

3.1.1 Task definition / Aufgabenbeschreibung

Das zu entwickelnde System soll Nutzergruppen mit einem berechtigten Interesse die Durchführung einer mittelspezifischen Risikoanalyse auf der Grundlage einer einheitlichen, fachlichen abgestimmten Berechnungsmethode und einheitlicher räumlicher Datenbasis ermöglichen. Das System erlaubt eine sogenannte georeferenzierte probabilistische Risikobewertung und stellt eine wichtige technische Komponente im zukünftigen Zulassungsverfahren für Pflanzenschutzmittel dar. Die Anwendung (System) soll sich dem Nutzer intuitiv erschließen. (AP2.1 [Anhang 1])

3.1.2 Theoretical background, site conditions, literature / Theoretische Grundlagen, Randbedingungen, Literaturhinweise

Das Programmverhalten orientiert sich an den in UAP2.1 (Anhang 1) dargestellten Funktionen. Die hierfür notwendigen theoretischen Grundlagen entsprechen den Vorgaben des Arbeitspakets 1 für das Standardszenario Graben und Arbeitspakets 3, „Hot-Spot Criteria 1000/10“ (chap. 3 und 6).

3.1.3 Units, formats, appreviations / Einheiten, Formate, Abkürzungen

In der Berechnungskomponente (vgl. Abbildung 1) gelten folgende Maßeinheiten/Formate:

Eingabe

- ppp (Pflanzenschutzmittel): Name des Pflanzenschutzmittels
- AI (Wirkstoff): Name des Wirkstoffes
- AR (Aufwandmenge): [g/ha] (Wirkstoff)
- RAC: [µg/l]
- Scenario: [Hops/Fruit/Vine]
- Date: [dd.mm.yyyy]

Metadaten

- Species: [Daphnia/Fish/Algae]
- DT50: [d]
- Application/Season: [1...>7] (Anzahl der Applikationen)
- Days between: [d] (Tage zwischen zwei Anwendungen):

Ausgabe

- Management Segments: [km] (der addierten Risikogewässerabschnitte)
- length: [m]
- tol. Effect: [%]

In der Simulationskomponente gelten folgende Maßeinheiten/Formate:

Eingabe

- Direction: [°] (Himmelsrichtung in Grad)
- Distance: [m]
- Shield vegetation: [ja/nein] (abschirmende Vegetation)

Ausgabe

- final load: [mg/m²]
- final concentration: [µg/l]

3.2 Functional hierarchy / Funktionshierarchie

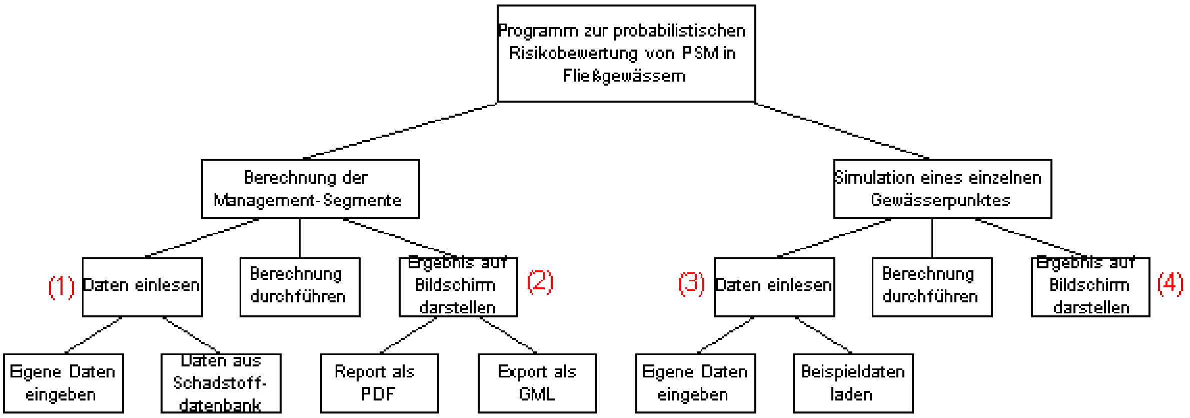


Abbildung 1: Funktionshierachie

3.3 Database tables / Datenbanktabellen

Die Beschreibung des Datenmodells erfolgt gemäß der Schemadatei (../database/uba08.jpg). Für eine optimale Übersicht der Datenbankrelationen empfiehlt sich nach dem Import der Datenbank (vgl. 5.2.2) ein Reverse-Engineering. Hierfür bietet Oracle kostenlose Tools bspw. den „JDeveloper“ an.

Im Kontext der Webbasierten Anwendung sind die folgenden Relationen explizit zu erwähnen:

Benutzer

Für die Anwendung werden verschiedene Datenbanktabellen benutzt. Zur Identifizierung des Benutzers wird die Tabelle „UBA08_USER“ benötigt.

Tabelle 1: Beschreibung der Benutzer-Tabelle

USERNAME	NAME	LASTNAME	ROLE	EMAIL	USERPASS
<VAR-CHAR2(20BYTE)>	<VAR-CHAR2(20BYTE)>	<VAR-CHAR2(20BYTE)>	<VAR-CHAR2(20BYTE)>	<VAR-CHAR2(20BYTE)>	<VAR-CHAR2(20BYTE)>

Der Wert für ROLE lautet entweder "ROLE_UBA" oder "ROLE_INDUSTRY" und entscheidet über unterschiedliche durch das Programm bereitgestellte Funktionalitäten. Ein Benutzer in der Rolle "ROLE_UBA" hat Zugriff auf die Schadstoffdatenbank.

Spezies

Für die Metainformation „Spezies“ wird eine Tabelle mit dem Namen „UBA08_SPECIES“ benötigt. In ihr wird der Name der Spezies, die Länge und Toleranzeffekt gespeichert.

Tabelle 2: Beschreibung der Spezies-Tabelle

ID	NAME	LENGTH	TOL_EFFECT
<NUMBER(38,0)>	<VAR-CHAR2(20BYTE)>	<NUMBER>	<NUMBER>

Pflanzenschutzmittel

Die Pflanzenschutzmittel sind in der Tabelle „UBA08_ACTIVE_COMPONENT“ mit folgendem Aufbau erfasst:

Tabelle 3: Beschreibung der Pflanzenschutzmittel-Tabelle

ID	NAME
<NUMBER(38,0)>	<VARCHAR2(20BYTE)>

Ein Pflanzenschutzmittel hat eine 1 zu n-Beziehung zu der Tabelle „UBA08_COMPONENT_TRIGGER“, in die Wirkstoffe mit ihren unterschiedlichen Ausprägungen vorliegen:

Tabelle 4: Beschreibung der Wirkstofftabelle

ID	NAME	COMPONENT_REF	SPECIES_REF	ERC	CROP	AI_AR
<NUMBER(38,0)>	<VARCHAR2(20BYTE)>	<NUMBER(38,0)>	<NUMBER(38,0)>	<NUMBER>	<VARCHAR2(20BYTE)>	<NUMBER(38,0)>

3.4 Error Handling / Fehlerbehandlung

Fehler können während des Betriebsauflaufes durch Benutzereingaben oder durch das System entstehen.

Benutzereingaben

Fehler und ungültige Werte bei der Eingabe werden unmittelbar in der Benutzeroberfläche angezeigt. Ein Weiterarbeiten ist nur möglich, wenn fehlende oder fehlerhafte Eingaben korrigiert wurden.

Ausgelöst durch System

Nach umfangreichen Testläufen sollten während der Datenbankabfrage oder der Simulation keine schweren Systemfehler auftreten. Tritt ein Fehler auf, könnte dies an fehlerhaften Datenbankeinträgen oder an einer unterbrochenen Datenbankverbindung liegen und es wird eine Fehlerseite angezeigt. Ein Neustart des Servlet-Containers oder ein (mehrfaches) Drücken des "Aktualisieren"-Buttons des Browserfensters könnten das Problem beheben. Ebenso kann ein „Hängenbleiben“ während der Berechnung mit einem Aktualisieren der Seite beseitigt werden.

Treten Fehler in der Simulationskomponente auf, wird in der Regel kein Neustart derWebapplikation benötigt.

4 Program organisation / Programmaufbau

4.1 Program structure, source code / Programmstruktur, Quellcode

Die Programmstruktur geht aus den Javadocs hervor. Die Quelltexte sind in den Klassendateien zu finden.

- Pfad zur Javadoc: /doc/*.html
- Pfad zu den Klassendateien: /src/*.java

4.2 Compiling / Kompilieren

Das System wurde mit MyEclipse Version 7.5 entwickelt. Das Projekt kann sich mit dieser Entwicklungsumgebung öffnen und kompilieren lassen. Die Projekt- sowie Quelldateien und Libraries sind auf der CD zu finden.

5 Installation

5.1 Requirements concerning devices, hard- and software / Gerätebedarf, Hard- und Softwarebedarf

Softwarevoraussetzungen (vgl. AP2.1, Anhang 1):

- Servletcontainer (Tomcat 6.x).
- Datenbank (Oracle Database 10g Release 10.2.x oder höher)

Hardwarevoraussetzungen: (internetfähiger Server in einer abgesicherten DMZ)

Der Server muss den o.g. Ansprüchen an die Software erfüllen. Es müssen mindestens 2 GB RAM für die Anwendung zugesichert werden. Datenbank und Webanwendung können physisch auch getrennt sein.

5.2 Program installation and configuration guidance / Programminstalation-sanweisung,-konfiguration

5.2.1 Deflate the program / Entpacken der Anwendung

Die Applikation liegt als .war-Archiv vor. Zur Dokumentation dieses Archiv sei auf die Originaldokumentation verwiesen.

http://java.sun.com/j2se/1.4.2/docs/guide/jar/jar.html

Dieses Archiv muss lediglich in das webapps-Verzeichnis des Tomcat deployed werden. Es ist darauf zu achten, dass sowohl ausreichende Schreibrechte als auch genügend Speicher für die virtuelle Maschine von Java (Java Heap Space) vorhanden sind, da die Applikation Daten temporär auf dem Server zwischenspeichert. Temporäre Daten sind Simulationsergebnisse in Form von Grafikdaten oder gml-Dateien.
Der Heap Space sollte mindestens 512 MB betragen. Dieser lässt sich z. B. mit dem Tomcat-Monitor (…/tomcatX/bin/tomcatXw.exe) unter dem Reiter „Java“ angeben.

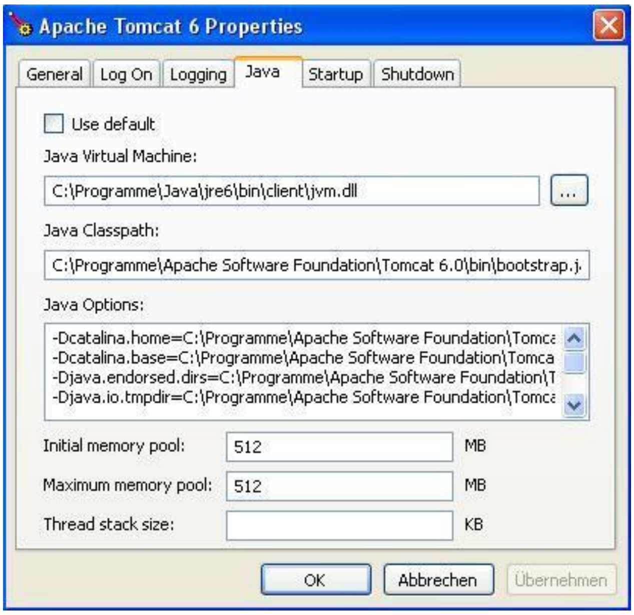


Abbildung 2: Angabe des Heap Space im Tomcat Monitor

5.2.2 Creating Tables in the data base / Anlagen der Tabellen in der Datenbank

Sämtliche von der Anwendung benötigte Tabellen befinden sich im oben genannten Datenbankdump. Der Datenbankdump (../database/uba08.zip) liegt als komprimiertes Archiv vor. Nach dem Entpacken des Archives kann die Datenbank als „Dump" in Oracle 10/11 importiert werden.

5.2.3 Connecting the application to the data base / Verbindung der Applikation zur Datenbank

Die Anbindung der Applikation an die Datenbank erfolgt durch eine XML- Parameterdatei. Dies geschieht in der Datei persistence.xml im Verzeichnis webapps/uba08web/WEB-INF/classes/META-INF. Diese muss den Netzwerkeinstellungen der Betriebsumgebung angepasst werden:

```
<persistence xmlns="http://java.sun.com/xml/ns/persistence"

       xmlns:xsi="http://www.w3.org/2001/XMLSchema-instance"

       xsi:schemaLocation="http://java.sun.com/xml/ns/persistence

    http://java.sun.com/xml/ns/persistence/persistence_1_0.xsd"

       version="1.0">

       <persistence-unit name="uba08webPU" transaction-type="RESOURCE_LOCAL">

              <provider>

                     oracle.toplink.essentials.PersistenceProvider

              </provider>
```

```
                <class>de.bund.jki.sf.uba08web.common.jpa.Uba08ActiveComponent</class>
                <class>de.bund.jki.sf.uba08web.common.jpa.Uba08ComponentTrigger</class>
                <class>de.bund.jki.sf.uba08web.common.jpa.Uba08User</class>
                <class>de.bund.jki.sf.uba08web.common.jpa.Uba08Species</class>
                <class>de.bund.jki.sf.uba08web.common.jpa.GeoRma</class>

                <properties>
                        <property name="toplink.jdbc.driver"
                                        value="oracle.jdbc.driver.OracleDriver" />
                        <property name="toplink.jdbc.url"
                                value="jdbc:oracle:thin:@XXX.XXX.XXX.XXX:1521:geodb" />
                        <property name="toplink.jdbc.user" value="benutzername" />
                        <property name="toplink.jdbc.password" value="benutzerpasswort" />
                </properties>
        </persistence-unit>
</persistence>
```

Abbildung 3: Parametereintrag zur Datenbankanbindung

Zu beachten ist, dass die Applikation ständig Verbindungen zu der Datenbank offen hält. Liegen Datenbank und Servlet-Container physisch getrennt vor, ist darauf zu achten, dass die Verbindungen nicht eventuell von einer Firewall getrennt werden. Es wird ein vom Servlet-Container verwalteter „Datenbank Verbindungspool“ empfohlen.

Es wird davon ausgegangen, dass eine Oracle 10g Datenbank oder höher verwendet wird. Soll eine andere Datenbank verwendet werden, muss lediglich die persistence.xml entsprechend den o. g. Empfehlungen angepasst werden. Das Verzeichnis webapps/uba08web/WEB-INF/lib muss um den spezifischen Datenbanktreiber ergänzt werden. Ein Reverse Engineering der Tabellenstruktur ist durch den JPA Standard möglich. Zur Dokumentation dieses Standards sei auf die Originaldokumentation verwiesen.

http://java.sun.com/javaee/reference/faq/persistence.jsp

5.2.4 Configuration and internationalisation / Konfiguration und Internationalisierung

Im Verzeichnis

uba08web/WEB INF/classes/de/bund/jki/sf/uba08web/common/properties

befindet sich die Datei "application_de.properties". Dies ist eine Textdatei, in der die Label, Buttons, Tabs, usw. sowie einige Parameter der Anwendung aufgeführt und parametrisiert werden.

```
#Main
button_logout=Logout
logged=Logged in as
#Tabs
tab_description=Project Description
tab_calculation=Calculation
tab_simulation=Simulation
tab_help=Help
tab_news=News
tab_about=About
...
```

Abbildung 4: Beispielauszug der application.properties

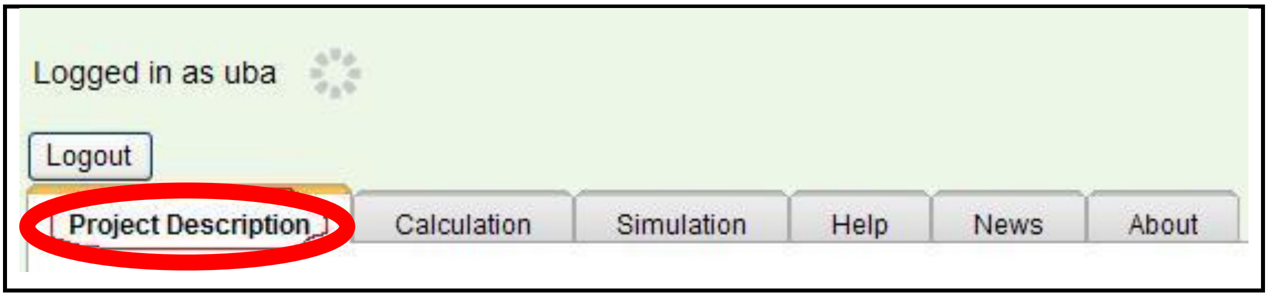


Abbildung 5: Beispiel der Internationalisierung der "Reiter/Tabs"

Abhängig von der Spracheinstellung des Browsers wird die properties-Datei mit der entsprechenden Endung (_de, _en, _fr, ...) benutzt. Ist eine Internationalisierung gewünscht, muss eine neue properties-Datei nach Vorbild der vorhandenen Datei mit der entsprechenden Länderkennung erstellt werden. Diese muss dann zusammen mit der Datei "application_de.properties" in dem properties-Verzeichnis liegen. In der properties-Datei lassen sich auch die Inhalte der Tabs „news", „help" und „about" angeben. Nach diesen Anpassungen muss der Servletcontainer neu gestartet werden. Zur Dokumentation dieses Standards zur Internationalisierung sei auf die Originaldokumentation verwiesen.

http://docs.sun.com/app/docs/doc/819-3669/bnaxu?a=view

5.2.5 Changing the e-mail address for registration / Änderung der Empfängeradresse für die Registrierung

Der Benutzer registriert sich über ein Formular. Hier gibt er Namen, Vornamen, E-Mail und die gewünschte Rolle an. Der Administrator erhält daraufhin eine Nachricht, die die Nutzerdaten beinhaltet.

Die E-Mail-Adresse des Administrators lässt sich in der Datei uba08web/WEB-INF/classes/de/bund/jki/sf/uba08web/common/properties/mail_de.properties angeben.

Hier kann auch der Textinhalt und der Betreff der Nachricht verändert werden.

Da sich die Zugehörigkeit eines Nutzers zu der Rolle Industrie oder UBA nicht automatisch klären lässt, muss der Administrator entscheiden und den Benutzer manuell der Datenbank hinzufügen.

5.2.6 Changing the layout of the report (optional) / Ändern des layouts des Reports (Optional)

Ein Report wird im pdf-Format erstellt und kann vom Benutzer heruntergeladen werden. Das Layout wurde mit JasperReports(www...) erstellt. Dieses lässt sich beliebig verändern. Die jxml-Datei liegt im Verzeichnis webapps/uba08web/jasper.

5.3 Running the Programm / Programmbetrieb

5.3.1 User manual / Bedinungsanweisung

Bei standardkonformer Installation des war-Archives (vgl. SUN- Spezifikation) lässt sich die Applikation über die Adresse <serveradresse>/uba08web erreichen.

Zuerst wird der Benutzer aufgefordert seinen Nutzernamen und sein Passwort einzugeben. Nach dem Drücken des Login-Buttons gelangt er auf die Hauptseite.

Über die Hauptseite sind folgende Menüpunkte zu erreichen:

Menüpunkt			Funktion
Project Description			Beschreibung des Projektes. Der Inhalt des Tabs kann in der properties-Datei unter dem Label "text_description" geändert werden.
Calculation			Unter diesem Menüpunkt findet sich die probabilistische Risikobewertung.
	Exposure Input		Hier werden die Daten sowie Metadaten, die zur Berechnung nötig sind, eingegeben.
		start	Startet die Berechnung
	Results PEC		Nach der Berechnung werden in diesem Tab die Ergebnisse ausgegeben. Der Benutzer hat hier die Möglichkeit einen Report der Berechnungsergebnisse zu erstellen und zu speichern.

		Download pdf	Lädt die Berechnungsergebnisse in einem pdf-Report herunter.
	Export RMS		Es werden GML-Dateien zum Export angeboten
		export…	RRGs lassen sich als Punktdaten im GML-Format herunterladen.
Simulation			Unter "Simulation" befindet sich die Simulationskomponente, mit der sich die zu erwartende PSM-Belastung eines einzelnen Gewässerpunktes berechnen lässt.
		calculation	Startet die Simulation
Help			Hilfe zu der Applikation. Der Hilfetext lässt sich in der properties-Datei unter dem Label "`text_help`" anpassen.
News			Neuigkeiten. Der dazugehörige Text findet sich unter"`text_news`".
About			Informationen über das Projekt. Anpassungen unter "`text_about`" in der properties-Datei.
Userdata			An dieser Stelle können die in der Datenbank gespeicherten Benutzerdaten verändert werden.

Unter dem Menüpunkt "Calculation / Exposure Input" lassen sich die Kenndaten des zu berechnenden PSM eingeben:

Eingabe	Typ	Zulässiger Bereich	Darf leer bleiben
ppp(plant protection product)	String	-	Nein
AI(Active Ingredient)	String	-	Nein
AR(Application Rate)	Double	>=0	Nein
Scenario	String	Hops, Fruit, Vine	Nein
Species	String	Daphnia,Fish,Algae	Nein
Dt50	Double	-	Nein
Application/season	String	12,3,...,n,>n	Nein
Days between	Integer	-	Nein

Für Double-Werte ist ein Punkt als Dezimaltrennzeichen vorgesehen.

Ist der Benutzer in der Rolle "uba" angemeldet, hat er Zugriff auf eine beispielhafte Schadstoffdatenbank.

Gibt der Nutzer ein in der Datenbank bekanntes Pflanzenschutzmittelprodukt in das Feld „ppp“ ein, so werden ihm in einer nebenstehenden Tabelle Wirkstoffe zur Auswahl angeboten. Deren Werte (AR, RAC, Scenario) können für die Berechnung übernommen werden. Diese Werte kann der Anwender auch nachträglich ändern.

Unter dem Tab „Simulation“ gibt der Benutzer die nötigen Daten zur Berechnung der Belastung eines einzelnen Gewässerpunktes ein.

Eingabe	Typ	Zulässiger Bereich	Darf leer bleiben
Distance[m]	Double	0-150	Nein
Percentile	Integer	0-100	Nein
Day of year	Integer	0-365	Nein
Shield vegetation	Boolean	true/false	Nein
Anzahl Simulationen	Integer	500-10000	Nein

6 Example / Anwendungsbeispiel

Calculation

Die Applikation wird durch die Eingabe "<serveradresse>/uba08web" in der Adressleiste des Browsers aufgerufen. Der Benutzer wird nach seinem Benutzernamen und –passwort gefragt.

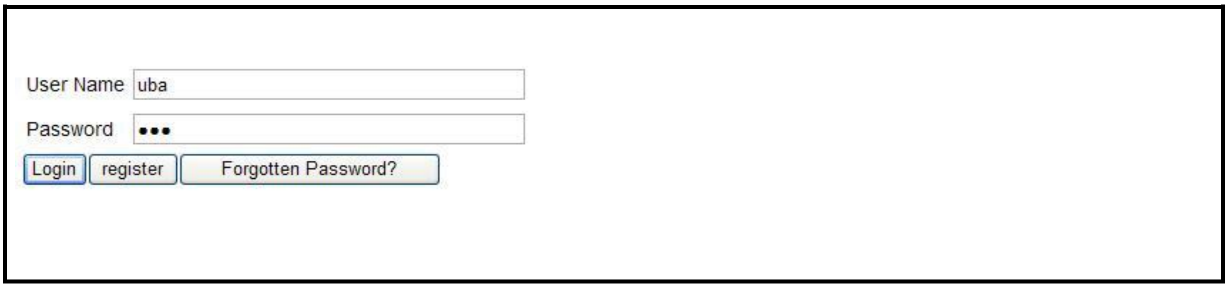


Abbildung 6: Login

Nach dem erfolgreichen Login kann der Benutzer unter dem Menüpunkt „Calculation" berechnungsrelevante Daten eingeben. Ist der Benutzer in der Rolle „uba" angemeldet, schlägt ihm das System bei der Eingabe des Namens des PSM Wirkstoffe aus der Datenbank vor.

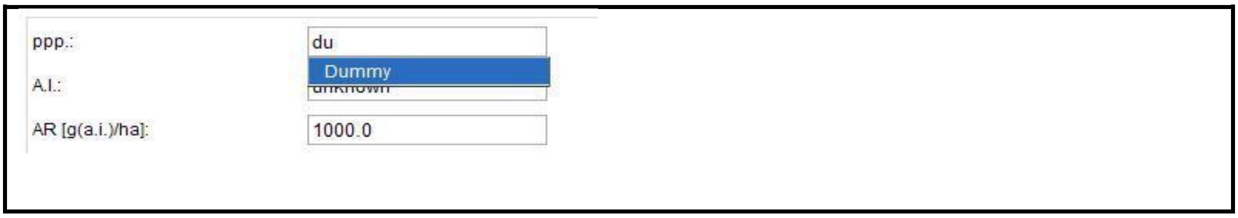


Abbildung 7: Autovervollständigung bei der Auswahl eines PSM

Die Wirkstoffe erscheinen in einer Tabelle, in der diese ausgewählt werden können. Die Werte der Wirkstoffe werden nach der Auswahl in den Eingabefeldern übernommen.

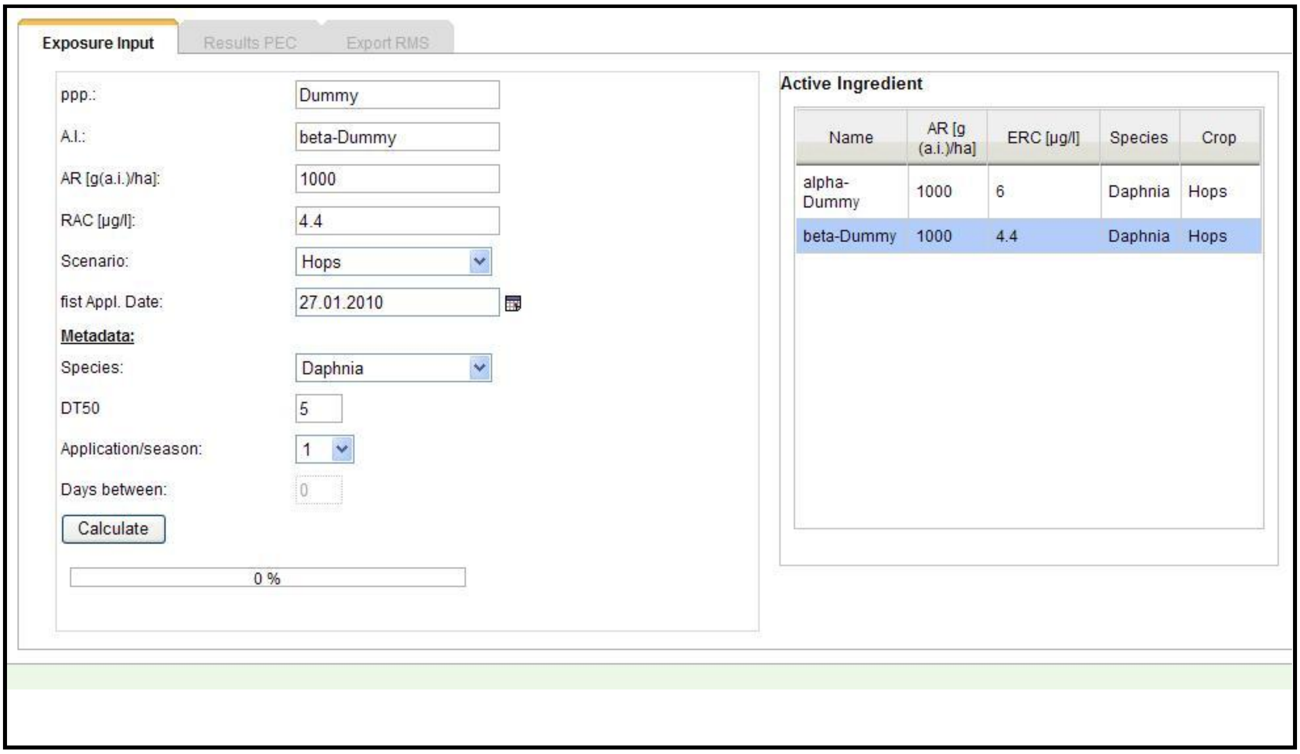
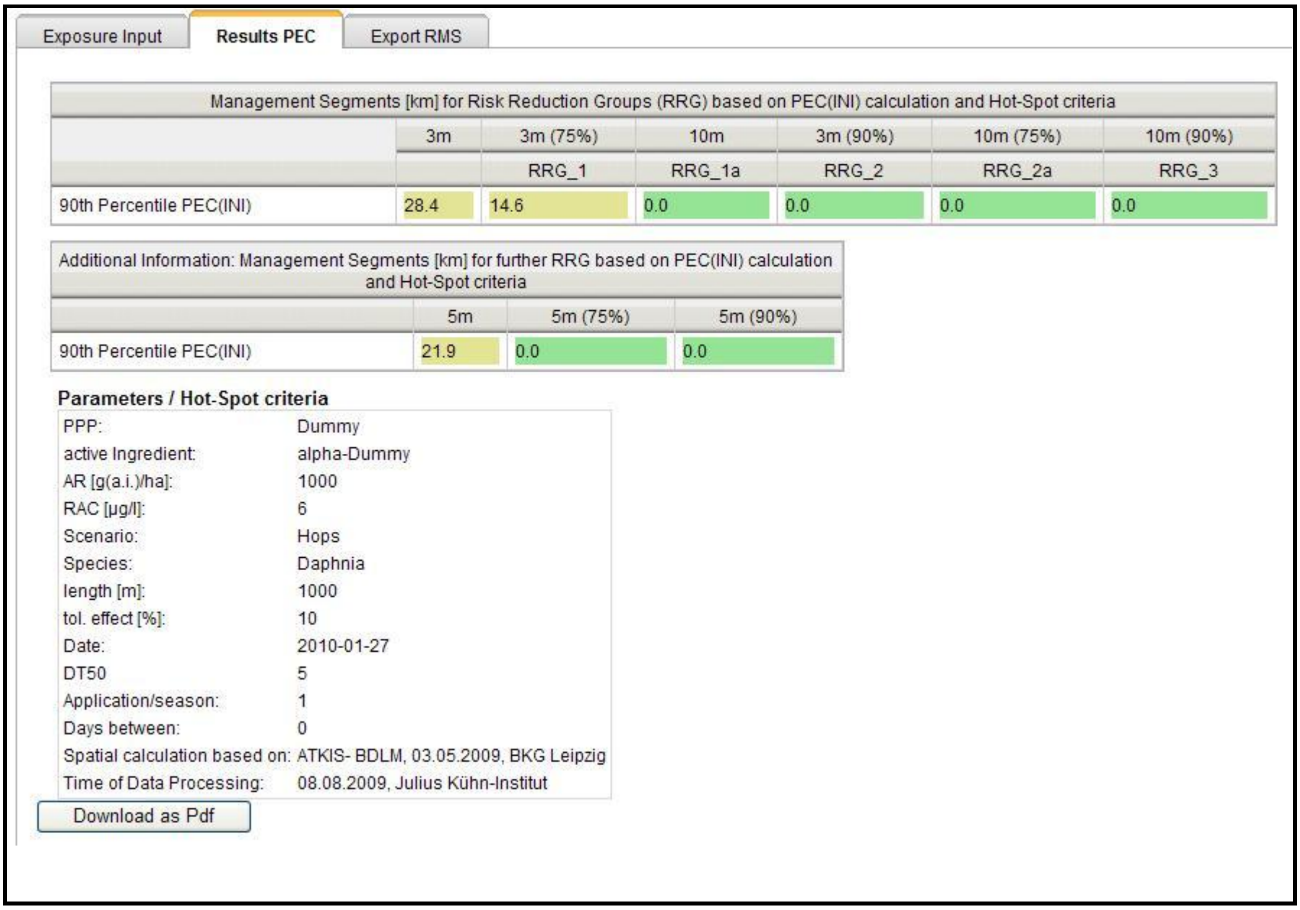


Abbildung 8: Ergebnis aus der Wirkstoffdatenbank

Nach Eingabe der Daten lässt sich die Berechnung mit dem Button „Calculate“ starten. Der Fortschrittsbalken zeigt den Stand der Datenbankabfrage. Ist diese beendet, wird der Tab „Results PEC“ aktiviert. Hier werden die Ergebnisse aufgeschlüsselt nach RRGs angezeigt. Die Ergebnisse geben Auskunft darüber, wie viel km der überprüften Gewässer potentiell gefährdet sind. Darüber hinaus erscheinen noch einmal die eingegebenen Werte als Metadaten.



Exposure Input | Results PEC | Export RMS

Management Segments [km] for Risk Reduction Groups (RRG) based on PEC(INI) calculation and Hot-Spot criteria						
	3m	3m (75%)	10m	3m (90%)	10m (75%)	10m (90%)
		RRG_1	RRG_1a	RRG_2	RRG_2a	RRG_3
90th Percentile PEC(INI)	28.4	14.6	0.0	0.0	0.0	0.0

Additional Information: Management Segments [km] for further RRG based on PEC(INI) calculation and Hot-Spot criteria			
	5m	5m (75%)	5m (90%)
90th Percentile PEC(INI)	21.9	0.0	0.0

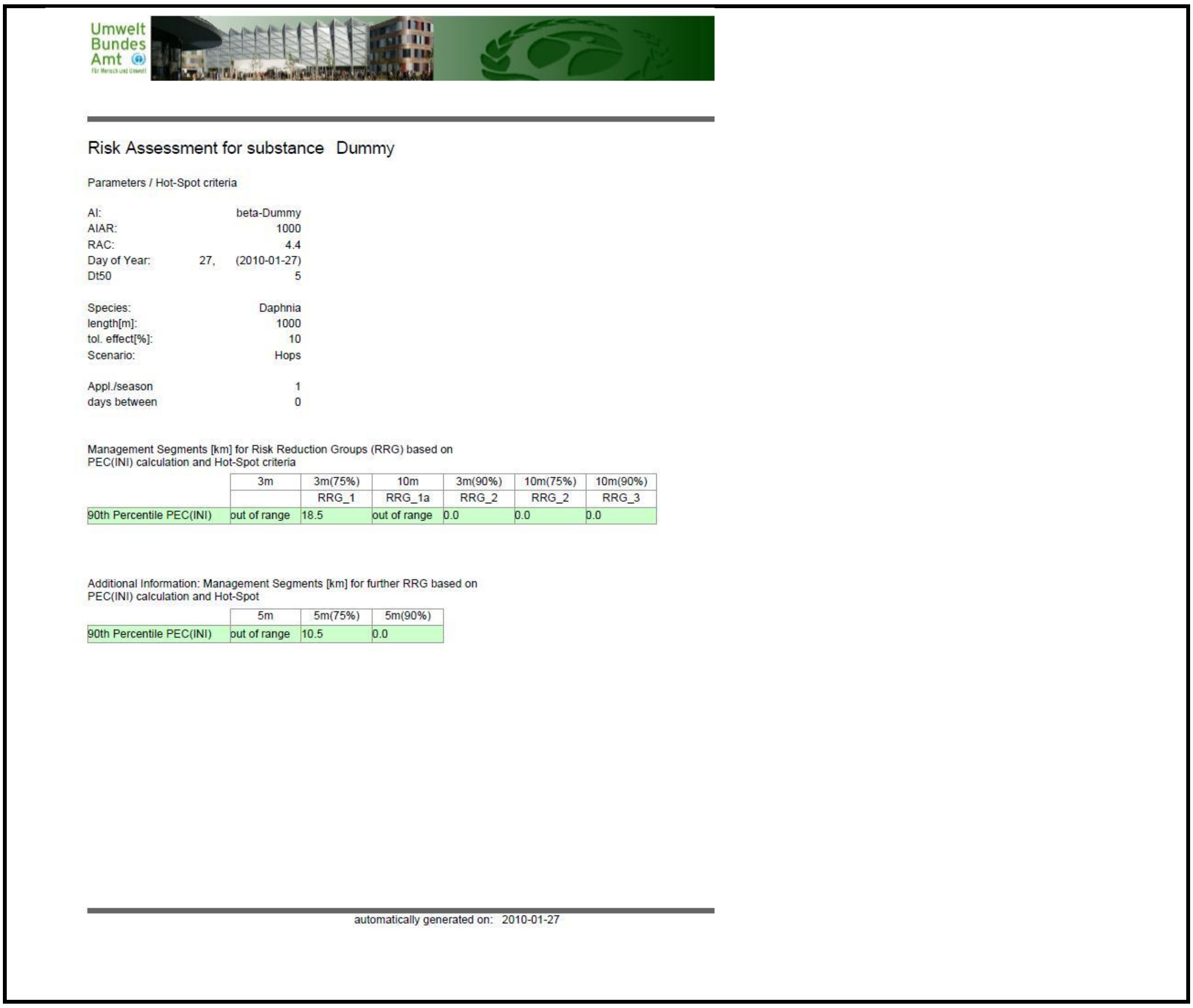
Parameters / Hot-Spot criteria

PPP:	Dummy
active Ingredient:	alpha-Dummy
AR [g(a.i.)/ha]:	1000
RAC [µg/l]:	6
Scenario:	Hops
Species:	Daphnia
length [m]:	1000
tol. effect [%]:	10
Date:	2010-01-27
DT50	5
Application/season:	1
Days between:	0
Spatial calculation based on:	ATKIS- BDLM, 03.05.2009, BKG Leipzig
Time of Data Processing:	08.08.2009, Julius Kühn-Institut

Download as Pdf

Abbildung 9: Darstellung der Simulationsergebnisse

Mit „Download as Pdf“ ist es möglich einen Bericht mit den Ergebnissen herunter zu laden.



Umwelt Bundes Amt

Risk Assessment for substance Dummy

Parameters / Hot-Spot criteria

AI:		beta-Dummy
AIAR:		1000
RAC:		4.4
Day of Year:	27,	(2010-01-27)
Dt50		5
Species:		Daphnia
length[m]:		1000
tol. effect[%]:		10
Scenario:		Hops
Appl./season		1
days between		0

Management Segments [km] for Risk Reduction Groups (RRG) based on PEC(INI) calculation and Hot-Spot criteria

	3m	3m(75%)	10m	3m(90%)	10m(75%)	10m(90%)
		RRG_1	RRG_1a	RRG_2	RRG_2	RRG_3
90th Percentile PEC(INI)	out of range	18.5	out of range	0.0	0.0	0.0

Additional Information: Management Segments [km] for further RRG based on PEC(INI) calculation and Hot-Spot

	5m	5m(75%)	5m(90%)
90th Percentile PEC(INI)	out of range	10.5	0.0

automatically generated on: 2010-01-27

Abbildung 10: Reportgeneration als *.pdf

Ist die genaue Lage der potentiell gefährdeten Gewässerabschnitte von Interesse, lässt sich unter dem Menüpunkt „Export RMS“ für jede einzelne RRG, in der ein Ergebnis vorliegt, eine GML-Datei herunterladen.

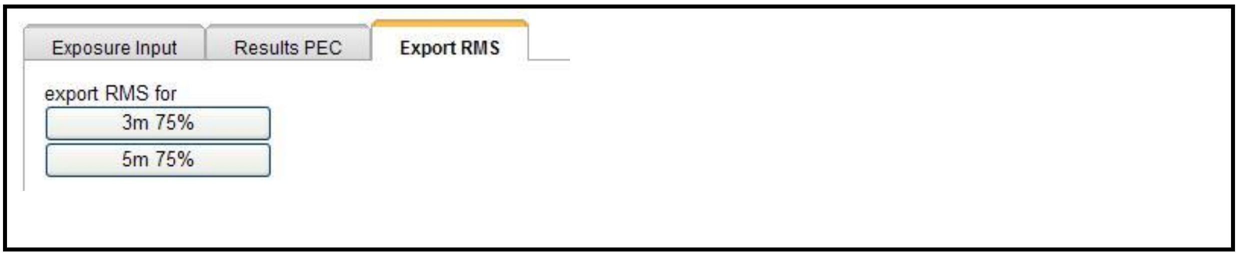


Abbildung 11: Export der MS als GML

Einzelsimulation

Mit der Simulationskomponente lassen sich für einen einzelnen Gewässerpunkt die Belastung und Konzentration eines Gewässerpunktes relativ zu angrenzenden Applikationsflächen simulieren. Der Benutzer gibt für acht Himmelsrichtungen die Entfernung zur angrenzenden Applikationsfläche an, bestimmt ein Perzentil, das Szenario, den Tag des Jahres und die Anzahl der Simulationen. Die Einzelsimulation spiegelt den momentanen Wissensstand des Forschungsprojektes hinsichtlich der relevanten Simulationsparameter wider.

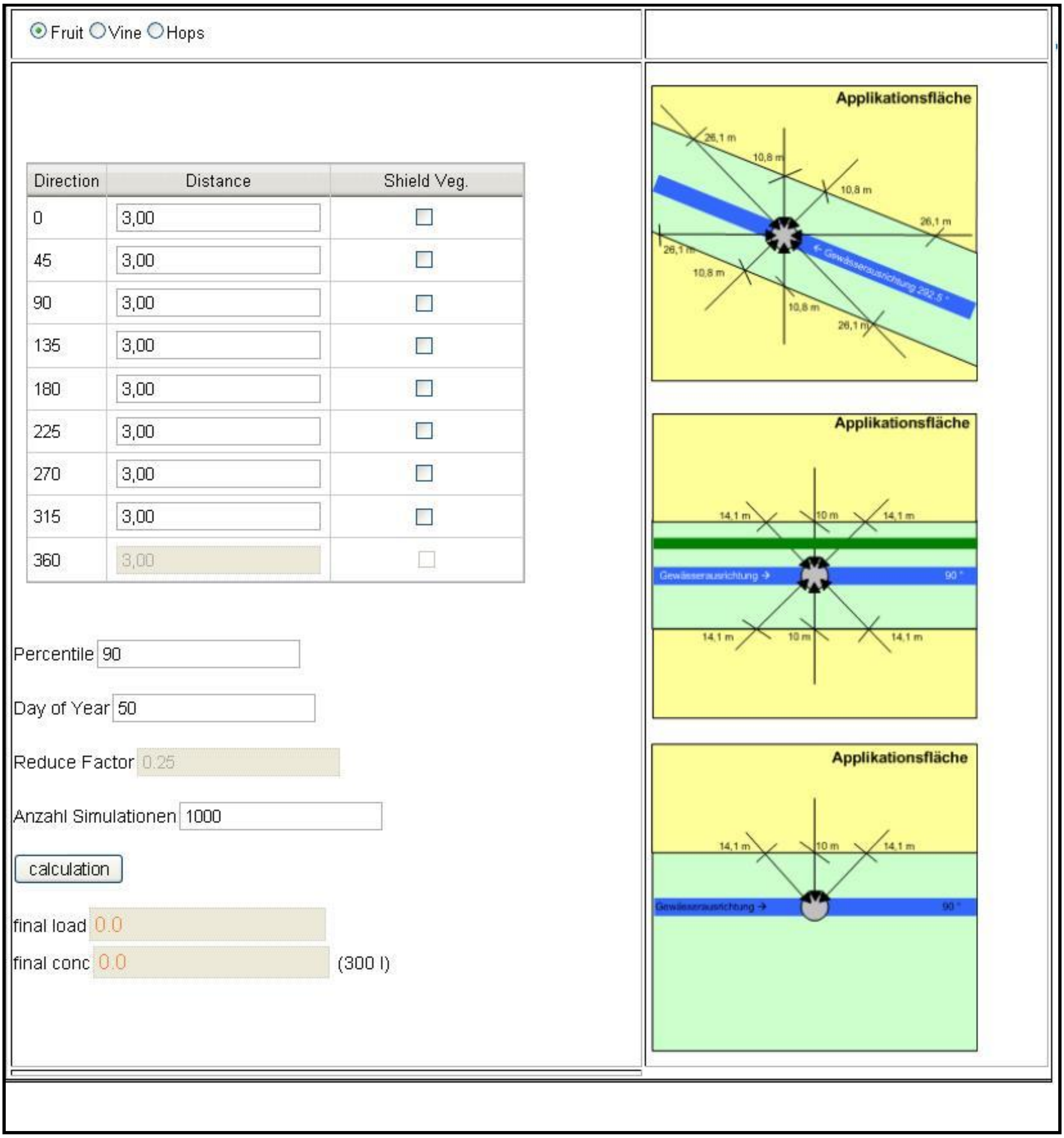


Abbildung 12: Anwendung der Einzelsimulation

Alternativ kann der Benutzer auf beispielhafte Eingabewerte zurückgreifen, indem er auf eines der drei nebenstehenden Bilder klickt. Die entsprechenden Werte werden dann in der Eingabemaske übernommen.

Direction	Distance	Shield Veg.
0	26,10	
45	10,80	
90	10,80	
135	26,10	
180	26,10	
225	10,80	
270	10,80	
315	26,10	
360	-1,00	

Abbildung 13: Beispielwerte eines selektierten Szenario

Nachdem die Berechnung beendet wurde, wird neben der final load und der final conc des Gewässerpunktes ein Diagramm dargestellt.

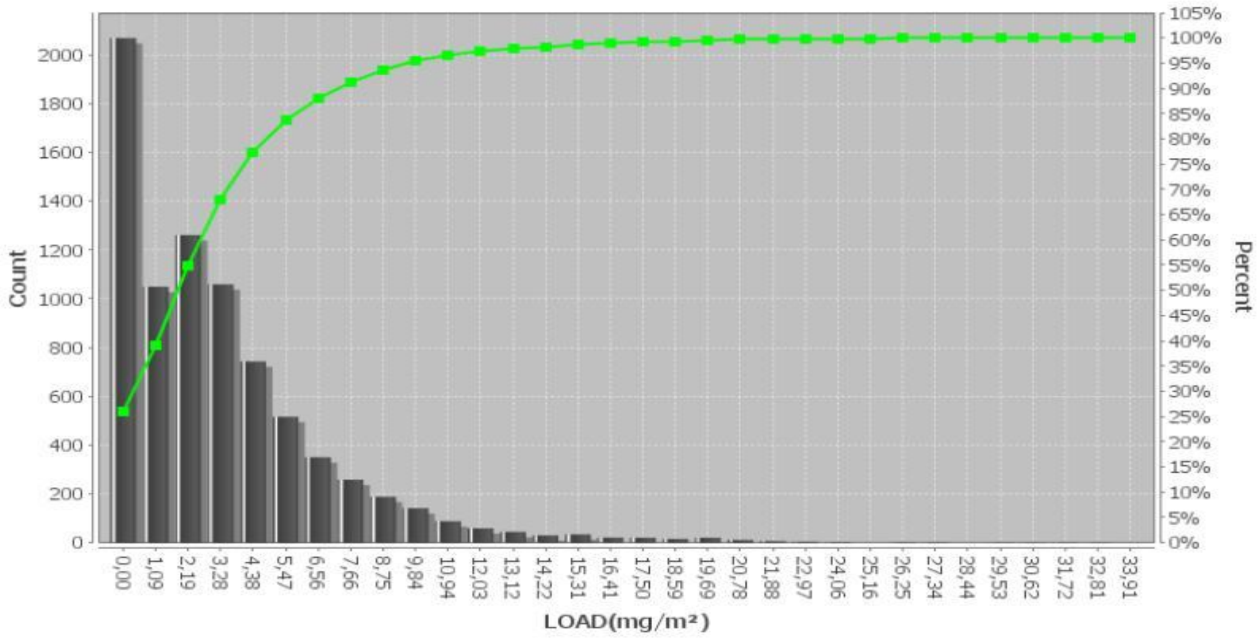


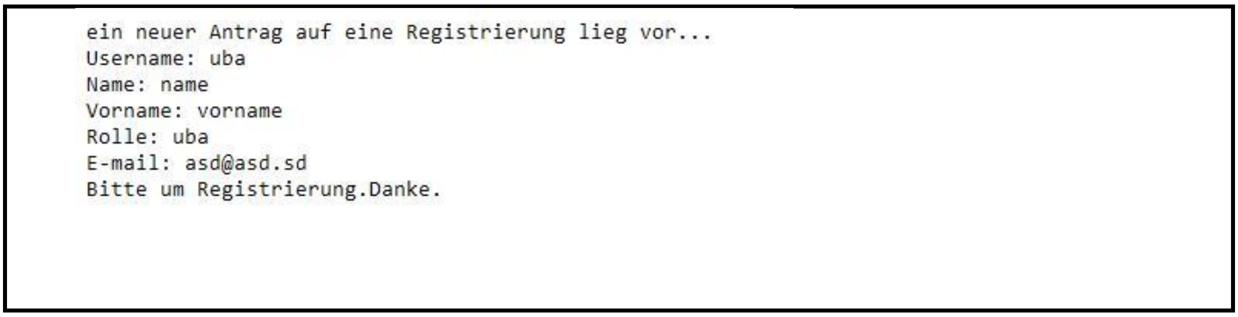
Abbildung 14: Ergebnis einer Einzelsimulation als Verteilung

Neu anmelden

Möchte sich ein neuer Benutzer registrieren lassen, wählt er auf der Login-Seite den Button „Register“.

Abbildung 15: Dialog für eine Neuanmeldung/Registrierung

Hier gibt er seine gewünschten Daten ein. Mit dem Drücken des submit-Buttons wird eine Email an die in der Datei „mail.properties“ angegebene Adresse gesendet:



```
ein neuer Antrag auf eine Registrierung lieg vor...
Username: uba
Name: name
Vorname: vorname
Rolle: uba
E-mail: asd@asd.sd
Bitte um Registrierung.Danke.
```

Abbildung 16: Textinhalt der Email

Der Benutzer muss nun manuell in die Tabelle „UBA08_USER“ eingetragen und über dessen Registrierung informiert werden.

7 File structure / Dateistruktur

Die Dateien auf der CD sind folgendermaßen angeordnet:

<root>

<app>

<uba08web.war>

<database>

<uba08.zip>

<doc>

<lib>

<src>

<techdoc.pdf>

8 Literature

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GeoRisk

Geodata based Probabilistic Risk Assessment of Plant Protection Products

(Georeferenzierte Probabilistische Risikobewertung von Pflanzenschutzmitteln)

UBA Project code 3707 63 4001, AZ: 93 112-5 / 9

Appendix C

Workshop Report

Georeferenzierte Probabilistische Risikobewertung von Pflanzenschutzmitteln:

GeoRisk

Authors

Udo Hommen, Roland Kubiak

April 23, 2010

UBA F+E Vorhaben 3707 63 4001, AZ: 93 112-5 / 9

Workshop

Georeferenzierte Probabilistische Risikobewertung von Pflanzenschutzmitteln:

GeoRisk

16. – 18. November 2009, Umweltbundesamt Dessau

Finaler Bericht, 23.04.2010

Autoren
Udo Hommen (Fraunhofer IME), Roland Kubiak (RLP AgroScience)

Fraunhofer IME

Justus-Liebig-Universität Giessen
Institut für Landschaftsökologie und Ressourcenmanagement

RLP AgroScience
Institut für Agrarökologie

JKI Julius Kühn-Institut
Institut für Strategien und Folgenabschätzung im Pflanzenschutz

Institut für Umweltforschung

gaiac
Forschungsinstitut für Ökosystemanalyse und -bewertung

RWTH Aachen

Inhalt

Zusammenfassung

Der aktuelle Stand des GeoRisk-Projekts (UBA F+E Vorhaben 3707 63 4001 sowie Optionen und Schwierigkeiten für eine Umsetzung wurde im November 2009 mit Vertretern verschiedener Interessensgruppen diskutiert. Der Fokus lag auf der Beurteilung der Risiken von Drifteinträgen aus Raumkulturen (Obst, Wein, Hopfen).

Vorgestellt wurden folgende GeoRisk-Projektergebnisse:

- Bundesweite Berechnung der PECsw (Predicted Envrionmental Concentration in surface water) auf der Basis von ATKIS und „Standardgrabenmodell“: Georeferenzierte probabilistische Berechnung der Einträge und zu erwartende km Managementsegmente je Kultur in Deutschland
- Unterschiede zwischen den PECs auf der Basis von ATKIS und High Resolution Data (HR sind größer in heterogenen, kleinräumiger strukturierten als in homogenen Landschaften
- Methode für Erzeugung eines topologisch korrekten, gerichteten Gewässernetzes, welches eine Vorraussetzung für eine dynamische Fließgewässermodellierung darstellt
- Untersuchung der sensitiven Parameter für die Ableitung von PECs in Fließgewässern
- Berechnung von Konzentrationsverläufen für Beispielgewässer unter konservativen Annahmen
- Entwicklung ökotoxikologischer Hotspot-Kriterien in Bezug auf Traitgruppen (im Aufbau)
- Konzept für Abschätzung der ökotoxikologischen Effekte kurzfristiger Exposition (kürzer als in Standardtests)
- Prototyp eines web-basiertes Anwendungstools zur Berechnung geroreferenzierter PECs und zur Identifiierung von Hotspots (räumliche Häufung kritischer Belastungen)

Die Hauptaussagen aus den Arbeitsgruppen und Plenardiskussionen des Workshops waren:

- Ziel eines georeferenzierten und probabilistischen Verfahrens zur Risikoabschätzung von Pflanzenschutzmitteln in D ist eine möglichst realitätsnahe Risikobewertung, auf deren Basis auch Vereinfachungen und Erleichterungen der bestehenden bundesweiten substanzbezogenen Anwendungsbestimmungen bei gleichzeitiger Gewährleistung eines ausreichend hohen Schutzniveaus möglich werden sollen..
- Das Schutzziel „keine unvertretbare Effekte auf lokale Populationen“ soll mit 95 % Sicherheit erreicht werden, wobei allgemein derartige Abschätzungen mit erheblichen Unsicherheiten behaftet sind. Dies bedeutet nicht, dass auf der Expositionsseite die 95. Perzentile einer PEC-Verteilung verwendet werden muss,. sondern dass mit 95%iger Sicherheit unvertretbare Effekte auf lokale Populationen bei Anwendung eines Pflanzenschutzmittels mit den entsprechenden Risikomanagementauflagen ausgeschlossen werden können. Auch beispielsweise eine 90. PEC-Perzentile kann zur gewünschten Sicherheit des Gesamtverfahrens beitragen.
- Die Unsicherheit in den Geodaten (Messfehler, Klassifizierungsfehler usw.) ist zu minimieren und alle Verfahrensunsicherheiten sind systematisch zu (Unsicherheitsanalyse) zu betrachten und die für die Risikobewertung relevanten Einflussparameter herauszustellen (Sensitivitätsanalyse).
- Die Berechnung von initialen PECs unter der Annahme des Standardgewässermodells (stehender Graben mit 30 cm Tiefe) ist als georeferenzierte probabilistische Berechnung des Drifteintrags zu betrachten. Für eine echte gerorefenrenzierte probabilistische Abschätzung der

Exposition wird der vorgestellte Ansatz eines dynamischen Modells, welches realistische Annahmen zur Gewässerstruktur (z.B. Breite-Tiefe-Verhältnis) sowie zu Transport und Dispersion in Fließgewässern berücksichtigen kann, als zielführend angesehen. Bis hin zur bundesweiten Anwendung besteht noch erheblicher Arbeits- und Entwicklungsaufwand.

– Eine (theoretische) Fallstudie ergab, dass entscheidende Einflussgrößen für die Expositionsbetrachtung im dynamischen Gewässermodell die Behandlungswahrscheinlichkeit und das Behandlungszeitfenster der Kulturflächen im Oberlauf eines betrachteten Segments sind. Ein Behandlungsfenster von 2 Arbeitstagen wurde vorläufig als eine plausible Annahme angesehen; zur genaueren Klärung wurde auf NEPTUN-Daten verwiesen.

– In Bezug auf vorgeschlagene Gleichverteilung der Windrichtungen bei Applikation und die Möglichkeit der Feststellung von lokalen Vorzugswindrichtungen in driftrelevanter Höhe über dem Boden wurde festgestellt, dass die wirkliche Verteilung der Windrichtungen bei Nutzung eines hohen Perzentils, z.B. des 90-Perzentils, der PEC-Verteilung am Gewässerpunkt keine Rolle spielt.

– Es bestand Einigkeit darüber, dass zwischen einer generischen substanzunabhängigen Analyse und der späteren (substanzspezifischen) Zulassungspraxis zu unterscheiden ist.

– In der generischen Risikoabschätzung soll eine sogenannte Hotspot-Analyse zur Identifizierung von Gewässerabschnitten mit einer ökologisch kritischen Häufung von Risikosegmenten unter Berücksichtigung des Wiedererholungs- und Wiederbesiedlungspotentials betroffener Populationen erfolgen.
Als Grundlage für die Hotspot-Kriterien dient vorläufig der vom UBA vorgeschlagene Ansatz (2007): Auf 1000 m Gewässerabschnitt soll in nicht mehr als 10 % der Abschnittslänge die vorhergesagte Konzentration (PEC) die Regulatorisch Akzeptierbare Konzentration (RAC, abgeleitet aus ökotoxikologischen Tests) überschreiten.

– Dieses konservative Kriterium soll im Projekt insofern verfeinert werden, dass Dosis-Wirkungsbeziehungen, kürzere Expositionsdauern als in den ökotoxikologischen Tests sowie die für verschiedene Arten unterschiedliche Höhe des tolerierbaren Effekts berücksichtigt werden können (Trait-Ansatz).

– Die Berücksichtigung von Hecken und ufernahe Krautvegetation als driftmindernde Faktoren kann auch bei der generischen Analyse kulturspezifisch erfolgen, wenn sichergestellt ist, dass Applikationen nur in der Jahreszeit mit voller Belaubung der Hecken oder starker Verkrautung der Gewässer erfolgen. Eine weitere Verfeinerung kann dagegen nur substanzabhängig erfolgen (z.B. Berücksichtigung des Verhaltens der Substanz im Gewässer).

– Die generische Risikoanalyse bietet die Grundlage für ein Risikomanagement in der Landschaft, in dem sie die Gewässersegmente identifiziert, an denen ökologisch kritische Einträge zu erwarten sind, welche durch Managementmaßnahmen verringert werden können.

– Art und Umfang des möglichen Risikomanagments in Hotspots bedarf noch der Klärung.

– Es bestand kein Konsens darüber, ob das Entstehen von (neuen) Hotspots das spätere Zulassungskriterium bilden soll, so wie es im GeoRisk-Ansatz vorgeschlagen wird. Vom IVA wurde vorgeschlagen, die generische Hotspot-Analyse und das sich daraus ergebende generische Risikomanagement wie in GeoRISK vorgeschlagen zu entwickeln, die Zulassungsentscheidung allerdings keinesfalls an einzelne Hotspots zu knüpfen.
Von BVL und IVA wurde vorgeschlagen, die Zulassungsentscheidung auf der Basis einer

bundesweiten PEC-Verteilung über alle (relevanten) Gewässersegmente zu treffen und sich von dem bisherigen „edge of the field"-Ansatz zu lösen.

- Es bestand kein Konsens über die notwendigen Entwicklungsschritte der benötigten Geodaten und deren zentrale Datenhaltung sowie die Entwicklung und Positionierung eines Tools (Modells) zur Expositions- und Risikobewertung.
  GeoRisk schlägt ein zentrales Tool vor: Für die Zulassungspraxis erfolgt durch die Behörde (UBA) die Erstellung der Geofachdaten- und Expositionsdatenbank, welche in gewissen Zyklen aktualisiert wird. Ein web-basiertes Tool erlaubt die Eingabe der substanzabhängigen Daten zu Anwendung und Toxizität. Eine Verfeinerung der Geofachdaten innerhalb eines einzelnen Zulassungsverfahrens wird vom UBA als nicht praktikabel angesehen und ist daher nicht vorgesehen. Dies kann nur im Rahmen der zyklischen Aktualisierung des Datenbestandes erfolgen und steht damit allen Antragstellern zur Verfügung.
- Das Projekt liefert Erkenntnisfortschritte, die laut PflSchG auch berücksichtigt werden sollten. Die Einführung eines neuen Verfahrens unter Berücksichtigung des Fließgewässeransatzes mit einer angepassten Bewertung der Kurzzeitexposition ist grundsätzlich möglich und wird von Workshopteilnehmern einem statischen Gewässermodell vorgezogen.
  Die Anwendung auf bundesweiter Ebene bedarf jedoch noch erheblicher Entwicklungs- und Arbeitsschritte. Die Anzahl der Managementsegmente (MS) liegt nach diesem Verfahren (dynamisches Modell) nach Einschätzung basierend auf einer Fallstudie unter denen des statischen Standardgrabenmodells. Genaue Aussagen zum Umfang des Risikomanagments in Hotspots sind aber noch nicht möglich.
- Die Workshopteilnehmer waren sich prinzipiell über die notwendigen nächsten Schritte einig:
  1. Abschluss des jetzigen Projektes
  2. Erarbeitung der notwendigen Parameter zur Bestimmung der Managementsegmente (MS) auf der Basis des Fließgewässermodells mit allen Beteiligten
  3. Berechnung der MS für ein Pilotgebiet (z.B. die Hallertau) und möglichst mindestens eines weiteren Gebietes mit anderer Kultur
  4. Auf dieser Basis Entscheidung über Etablierung eines Pilotprojektes
  5. Durchführung eines Pilotprojekts, z..B. Erprobung des Ansatzes in der Hallertau als klar definiertes und durch eine Kultur geprägtes Gebiet mit begleitendem chemischen und biologischen Monitoring
  6. Entscheidung über die Einführung des Verfahrens zunächst für alle Raumkulturen
  7. Insgesamt wurde der Wunsch nach einer aktiven koordinierenden Stelle für die Implementierung eines georeferenzierten probabilistischen Ansatzes geäußert, z.B. durch das BVL-Lenkungsgremium Probabilisitik

Bericht

Vom 16. bis zum 18.11.2009 fand im Umweltbundesamt in Dessau der GeoRisk Workshop statt, auf dem die aktuellen Ergebnisse des UBA-Projektes „Georeferenzierte Probabilistische Risikobewertung von Pflanzenschutzmitteln“, UBA Vorhaben 3707 63 4001, dem Projektgeber sowie Vertretern verschiedener Forschungsinstitute und Interessensgruppen (u.a. Pflanzenschutzmitteldienste, Bundesamt für Naturschutz, BVL, Industrie, UFZ Leipzig, Universitäten) vorgestellt wurden. In Arbeitsgruppensitzungen (Anlage 3) und Plenardiskussionen (Anlage 4) wurden anschließend Status, Vor- und Nachteile des Ansatzes sowie Optionen und Schwierigkeiten für eine Umsetzung diskutiert. Alle gezeigten Präsentationen sind in Anlage 5 zusammengestellt.

1. Tag

Zunächst wurden in Plenarvorträgen der Hintergrund und die Zielsetzung des Projektes vorgestellt.

Herr Matezki fasste den Stand zu Beginn des Projektes zusammen. Die Schlüsselergebnisse des Vorgängerprojektes der Universität Landau mit dem Workshop im Januar 2007 in Dessau (UBA FuE-Vorhaben, FKZ 206 63 402, Schulz et al. 2007[1]) aus UBA-Sicht waren:

1. Anerkennung der „end of tail“ bzw. „Hotspot“-Problematik in der Geo-PRA
2. Identifizierung ökologisch relevanter Häufungen (´biological traits´-Ansatz für ´Hotspot´-Definition)
3. Vierstufiges Verfahren der GeoPRA für Abdrift in Raumkulturen (einschl. RMA-Management)
4. Diskussion und Empfehlungen zur praktischen Umsetzung in Raumkulturen
5. Identifizierung offener Punkte z.B. zu Modellannahmen (Fließgewässer-Dynamik), Datenverfügbarkeit usw.

Charakteristisch für die vorgeschlagene Zulassungspraxis ist die Ableitung einer PEC-Verteilung in der Landschaft, aus der dann nach Auswahl eines Perzentils die bundesweiten Abstandsauflagen abgeleitet werden sollen. Der in diesem Projekt erarbeitete Verfahrensvorschlag ist in der folgenden Abbildung zusammengefasst.

[1] Schulz, R., Elsaesser, D., Ohliger, R., Stehle, S., Zenker, K. (2007), Umsetzung der georeferenzierten probabilistischen Risikobewertung in den Vollzug des PflSchG – Pilotphase – Dauerkulturen. Endbericht zum F & E Vorhaben 206 63 402 des Umweltbundesamtes, Institut für Umweltwissenschaften, Universität Koblenz-Landau, Landau, Germany, 129 p

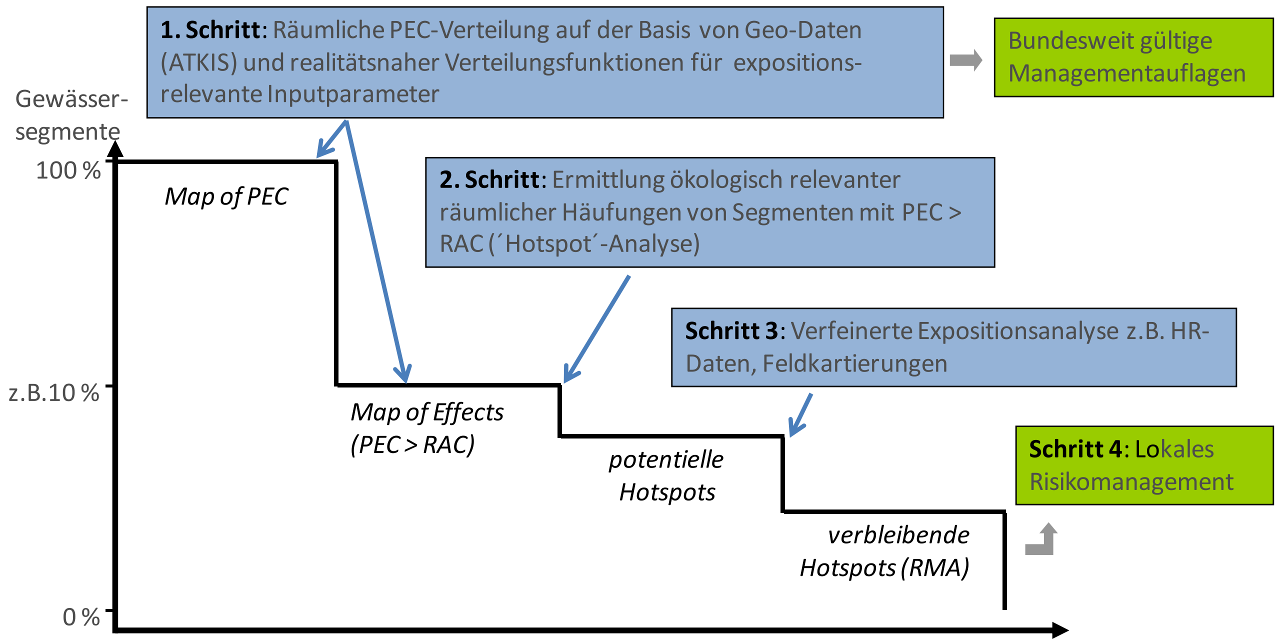

Abb. 1: Vorschlag eines 4-gestuften Ansatzes der Geo-PRA in Raumkulturen nach UBA-Projekt FKZ 206 63 402, Stand 2007), aus Präsentation von S. Matezki, s. Anhang 2

Udo Hommen gab danach einen Überblick über Ziele und Aufgaben des GeoRisk-Projekts und den bisher erarbeiteten Ansatz. Generelles Ziel ist durch eine realitätsnähere Risikobewertung die Grundlage für eine Vereinfachung und Erleichterung der bestehenden bundesweiten substanzbezogenen Anwendungsbestimmungen unter Gewährleistung eines ausreichend hohen Schutzniveaus zu schaffen; auch mit einem neuen Verfahren sollen mit 95 % Sicherheit regulatorisch nicht akzeptable Effekte auf aquatische Populationen ausgeschlossen werden können.

Im einzelnen soll in GeoRisk die wissenschaftliche Basis für die Einführung der georeferenzierten probabilistischen Bewertung bereitgestellt und die im vorhergehenden FuE-Vorhaben identifizierten offenen Punkte bzw. Datenlücken geklärt werden. Das Projekt beschränkt sich dabei auf Einträge von Pflanzenschutzmitteln in Gewässer durch Drift und Verflüchtigung[2] aus Dauerkulturen (Obst, Wein, Hopfen).

Fünf Grundelemente des GeoRisk-Ansatzes wurden vorgestellt:

1. Georeferenzierte und probabilistische Expositionsabschätzung
2. Berücksichtigung von Fließgewässern (dynamisches Modell)
3. Hotspot-Analyse: Identifizierung von Gewässerabschnitten mit ökologisch kritischer Häufung von Risikosegmenten unter Berücksichtigung des Wiedererholungs- und Wiederbesiedlungspotentials
4. Generische Risikobewertung zur Vorbereitung eines landschaftsbezogenes, substanzunabhängigen Risikomanagements, welches nach Umsetzung zur Vereinfachung und Erleichterung der substanzspezifischen bundesweiten Abstandsauflagen führt.

[2] Verflüchtigung ist in Arbeitspaket1 (theoretisch) behandelt, aber es gibt bisher keine generische Hotspot-Analyse für den Eintragspfad Verflüchtigung und Deposition. Es werden aber keine zusätzlichen Hotspots zu denen durch Drifteinträge erwartet.

5. Produktbezogene Bewertung mit Zulassungskriterium: „keine Hotspots in der Landschaft“, aus der sich ggf. mittel-spezifische bundesweite Abstandsauflagen ergeben

Der Unterschied zum Ansatz aus Schulz et al. 2007 (s. Abb. 1) liegt somit vor allem in der Entwicklung dynamischen Modells zur Beschreibung von Fließgewässern und im Vorschlag eines Zulassungskriteriums auf der Basis von Hotspots. Beiden Ansätzen ist die generische, substanzunabhängige Analyse zur Identifizierung potentieller Hotspots in der Landschaft als Basis für ein landschaftsbezogenes Risikomanagement gemein.

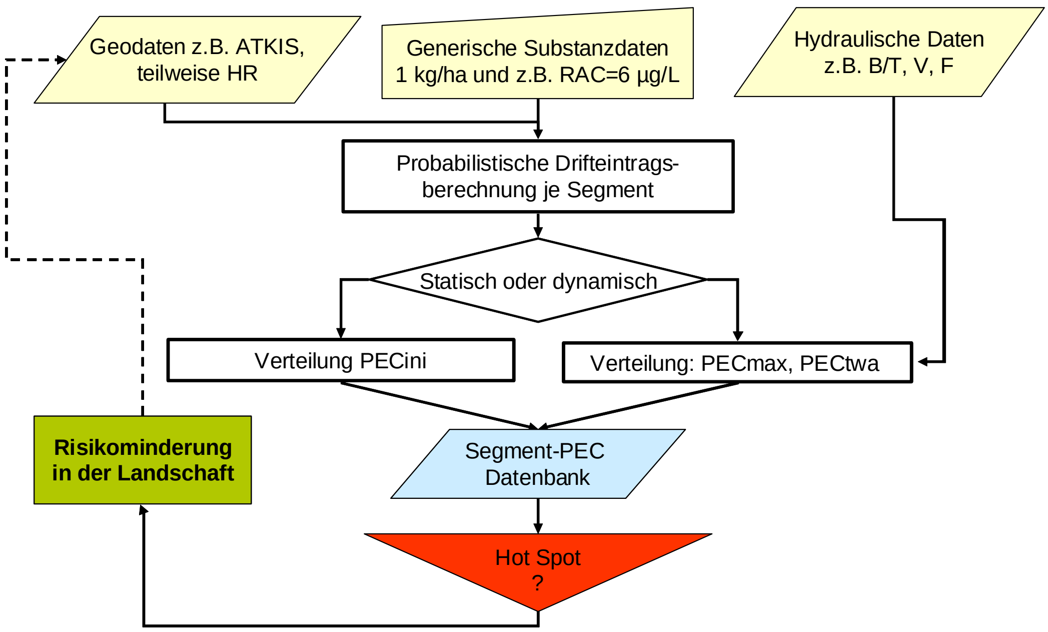

Abb. 2: Generische Risikoabschätzung zur Identifikation potentieller Hotspots in der Landschaft (aus Präsentation von U. Hommen, s. Anhang)

Das von GeoRisk vorgeschlagene zweistufige Zulassungsverfahren ist in Abb. 3 dargestellt.

Zur Vereinfachung ist die georeferenzierte probabilistische Berechung der PECs getrennt nach Fließ- und Stehgewässer nicht im Einzelnen dargestellt.

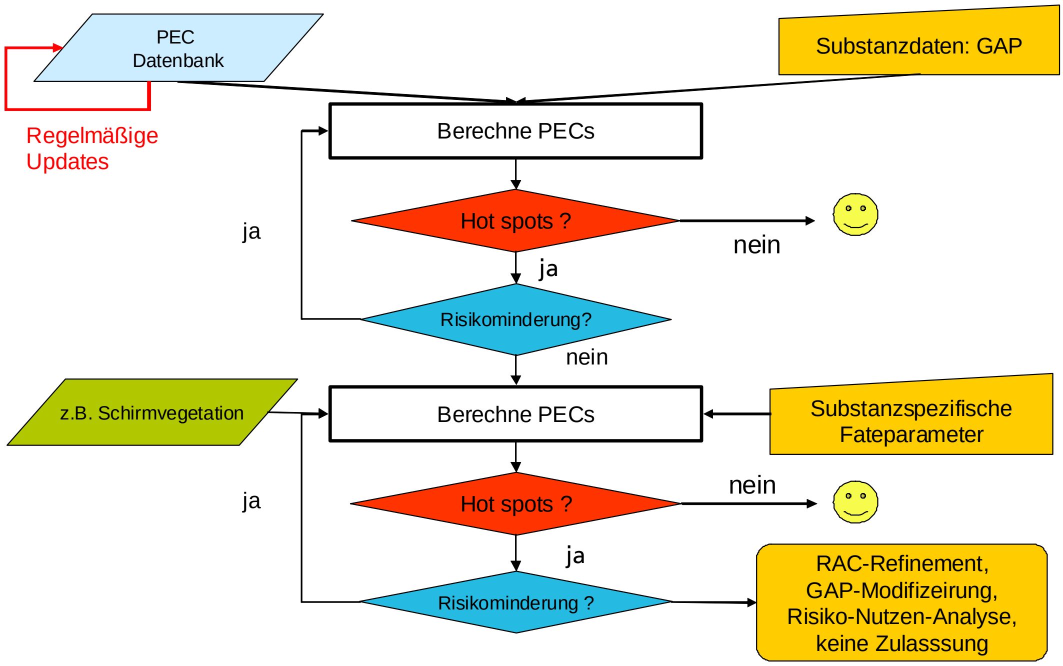

Abb. 3: Zulassungskonzept nach GeoRisk (aus Präsentation von U. Hommen, s. Anhang)

Die Chancen des GeoRisk-Ansatzes wurden vom Projektkonsortium in der Berücksichtigung von mehr Realität im Beurteilungsverfahren und der möglichen Vereinfachung und Erleichterung der bundesweiten Anwendungsbestimmungen bei Sicherung des Schutzniveaus gesehen. Als Herausforderungen wurden die Komplexität des Ansatzes (nicht unbedingt der Anwendung des Tools zur mittelspezifischen Risikoabschätzung) und die Umsetzbarkeit, insbesondere des landschaftsbezogenen Risikomanagements herausgestellt.

In einer Diskussion über Schutzziel und Schutzniveau wurde von Seiten des UBA erklärt, dass die Regulation auf der Basis des „Community Recovery Prinzips" erfolgen soll, d. h. es sollen langfristige unvertretbare Effekte auf lokale Populationen vermieden werden. Aus dem gesetzlichen Auftrag, dass „mit an Sicherheit grenzender Wahrscheinlichkeit" solche Auswirkungen auszuschließen sind, leitet sich eine angestrebte Wahrscheinlichkeit (Sicherheit) von 95 % ab. Diese angestrebte Sicherheit ist nicht mit einem bestimmten Perzentil einer PEC-Verteilung gleich zu setzen, da auch an anderer Stelle (z.B. bei der Effektbewertung und der Ableitung der Hotspot Kriterien) Annahmen einfließen, welche die Sicherheit der gesamten Risikoabschätzung beeinflussen. Bei der Bewertung der Sicherheit des Verfahrens sei natürlich auch zu berücksichtigen, dass die angenommene Wiedererholung von Populationen zum Zeitpunkt eines Eintrags nicht zu 100% gegeben ist, wenn bereits Effekte durch vorangegangene Anwendungen (Mehrfachbehandlung, Spritzserien) anzunehmen sind.

In den folgenden Vorträgen von M. Klein, B.Golla, M. Bach und M. Trapp wurden die Konzepte zur Eintrags- und PEC-Berechnung näher erläutert.

Michael Klein beschrieb wie die Drifteinträge in einzelne Segmente georeferenziert und probabilistisch berechet werden. Als georeferenzierte Parameter gehen der Abstand zwischen Kultur und Gewässer sowie das Vorhandensein von Drifteinträge verringernder Vegetation in die Berechnung ein. Probabilistisch werden die Depositionsraten in Abhängigkeit von der Entfernung auf der Basis der BBA-Driftversuche sowie die Windrichtung behandelt. Die Windgeschwindigkeit wird nicht explizit berücksichtigt, sondern als in den BBA-Versuchen abgebildet angenommen. Driftmindernde Technik und der Effekt driftmindernder Vegetation (Hecken, Uferrandvegetation) wird deterministisch wie in Exposit oder Focus berücksichtigt. Im Ergebnis wird für jedes Gewässersegment eine Verteilung der Einträge erzeugt, aus der dann ein bestimmtes Perzentil, z.B. das 80., 90, oder 95., für die PEC-Berechnung verwendet werden kann.

Burkhard Golla stellte die Anwendung dieses Modells auf der Basis von ATKIS-Daten und der Annahme des im jetzigen nationalen Verfahren verwendeten Standardgewässers (stehend, 1 m breit, 30 cm tief) für die Raumkulturen Hopfen, Obst und Wein in der Bundesrepublik und eine generische ‚worst case Substanz' vor. Zusammenfassend konnte Folgendes festgestellt werden:

Der Anteil der Risiko-Segmente (PEC > RAC) beträgt zwischen 15% (Hopfen, entspricht 146 km von 1000 km im driftrelevantem Bereich) und 20 % (Obstbau, Teilgebiet Bodensee, 137 km von 630 km im driftrelevantem Bereich).

Risikosegmente werden durch die Entfernung zwischen Applikationsfläche und Gewässer definiert (RS bei d < 10m). Die Anwendung des UBA Hotspot Kriteriums (10% PEC>RAC auf 1000m) führt zu einer sehr geringen Reduktion der RS Länge (< 5 %). In einem weiteren Vortrag gingen B: Golla und M. Trapp auf Unsicherheiten in den Geodaten ein, z.B. in Bezug auf die Abschirmung von Gewässern durch Hecken sowie die vollständige Erfassung der Gewässer und der Applikationsflächen. So konnte für Beispielgebiete gezeigt werden, dass ca. 86 % der Gewässer in ATKIS und Luftbild übereinstimmend erkannt werden. Ca.5 % der Gewässer, die im Luftbild erkannt werden, fehlen in ATKIS; umgekehrt können ca. 8 % der Gewässer nach ATKIS in den Luftbildern nicht gefunden werden. Für die Hallertau und das Obstgebiet Bodensee konnte an Beispielen gezeigt werden, dass die Anbaufläche in 3 bzw. 10 m Abstand zu Gewässern bei der Verwendung von HR-Daten meist erheblich unter der auf ATKIS-Basis berechneten Fläche liegt. Für Obstanbau in Baden bei 10 m Puffer sagten die HR-Daten jedoch einen höheren Anteil der Nutzflächen als ATKIS vorher. Generell waren Unterschiede von ATKIS und HR bei Betrachtung der definierten RAC und damit der Risikosegmentberechnung gering (je näher Applikationsflächen an Gewässern liegen, desto geringer war der Unterschied); ATKIS liefert daher mit hoher Sicherheit Hinweise auf Gewässer mit erhöhtem Risiko. Bei genauer Betrachtung der Verteilung aller PECini-Werte zeigten sich aber auch in intensiv genutzten Regionen Unterschiede.

Im nächsten Präsentationsblock wurde das dynamische Expositionsmodell für Fließgewässer vorgestellt. Martin Bach wies darauf hin, dass das Standardszenario eines stehenden Gewässers mit einem Breite/Tiefe-Verhältnis von 3.3 im Freiland eher die Ausnahme sei und daher für eine realistische Expositionsbetrachtung Gewässergeometrie und Fließverhalten zu berücksichtigen seien.

Die Modellierung der Gewässer als Transportsystem hat zur Konsequenz, dass einzelne Wasserpakete mehrfach Einträge erhalten können, dass Ort des Eintrages und der für den Effekt zu betrachtenden Ort nicht mehr identisch sind und dass für die Effektbewertung der zeitliche Verlauf der Exposition an einem Ort zu betrachten ist. In einer theoretischen Fallstudie wurde für die Beschreibung des Expositionsprofils an einem Ort die maximale PECtwa über 1 h sowie die Zeit, in der die PEC über der RAC liegt (Time over threshold), vorgestellt. Die Auswirkung des Expositionsmusters im Oberlauf (Zeitrahmen und Abfolge der Applikationen) wurde mit Hilfe von Beispielrechnungen vorgestellt. Für zwei konkrete Gewässer (Haunsbach bei Meilenhofen in der Hallertau und einem Gewässer bei Fischbach am Bodensee) zeigte Matthias Trapp Expositionsprofile unter der Annahme einer gleichzeitigen Behandlung aller Flächen.

Bach und Trapp stellten heraus, dass prinzipiell die HR-Landschaftsanalyse und Fließgewässermethodik deutschlandweit geleistet werden kann, aber sehr zeitintensiv ist und eine

Weiterentwicklung der bisher entwickelten Methodik noch notwendig ist. Möglich sei aber schon jetzt ein Higher Tier Ansatz nach der bundesweiten Hotspot-Berechnung mit dem statischen Gewässermodell auf der Basis von Szenarien für das Applikationsmuster.

Zusammenfassend wurde für den **Themenbereich Expositionsberechnung** herausgestellt und diskutiert, dass die Unsicherheit in den Geodaten (Lage, Abstände, Flächennutzung, …) zu minimieren sei (anders als bei Landschaftsperzentilen) und dass alle Verfahrensunsicherheiten systematisch zu betrachten seien.

Die vorgeschlagene Gleichverteilung der Windrichtungen bei Applikation und die Möglichkeit der Feststellung von lokalen Vorzugswindrichtungen in driftrelevanter Höhe über dem Boden wurden diskutiert. Von Seiten des Konsortiums wurde darauf hingewiesen, dass dazu keine quantitativen Daten zu erhalten wären und dass die wirkliche Verteilung der Windrichtungen bei Nutzung eines 90-Perzentils der PEC-Verteilung am Gewässerpunkt keine Rolle spiele (anders als beim Median oder Mittelwert).

Weiterhin wurde festgestellt, dass bei der generische Hotspot-Ermittlung neben der RAC-Ableitung auch die Berücksichtigung von Hecken und ufernaher Krautvegetation kulturspezifisch erfolgen kann (beispielsweise, wenn sichergestellt ist, dass Applikationen nur in der Jahreszeit mit voller Belaubung der Hecken oder starker Verkrautung der Gewässer erfolgen). Ein weiteres Refinement kann dagegen nur substanzabhängig erfolgen (z.B. Berücksichtigung des Verhaltens der Substanz im Gewässer).

Es wurde darauf hingewiesen, dass für die vorgestellten bundesweiten Risikoabschätzungen mit dem statischen Gewässermodell nur der Eintrag georeferenziert und probabilistisch abgeschätzt wurde, da für die PEC-Berechnung immer von der Geometrie eines Standardgrabens (30 cm Tiefe) ausgegangen wurde.

Im vorgestellten Fließgewässeransatz ist die Dispersion (Verdünnung) bisher ein relativ unwichtiger Prozess für die berechneten Expositionsprofile. Entscheidender für die berechneten maximalen PECtwa über eine Stunde und die Zeiten, in denen die RAC überschritten wird (Time over Threshold), sind die Behandlungswahrscheinlichkeit der Applikationsflächen im Oberlauf und das Behandlungszeitfenster.

Die Fliessgewässermodellierungen waren - so wie vorgestellt - noch relativ am Anfang. Gewässerseitig waren diese Modellierungen nach wie vor stark vereinfachend, indem konstante Gewässergeometrie und Dispersion über eine längere Fliesstrecke angenommen wurden. Es sollten hier weitergehende Modellbetrachtungen durchgeführt werden, bevor eine Bewertung stattfinden kann.

2. Tag

Roland Kubiak fasste zu Beginn des 2. Workshoptages die Kernaussagen des Vortages wie folgt zusammen:

- Aus Sicht des UBA sind 95 % Sicherheit für die Schutzzielerreichung „Keine unvertretbare Effekte auf lokale Populationen“ zu verlangen.
- Dies bedeutet nicht unbedingt, dass von PEC-Verteilungen dann auch die 95. Perzentilen genommen werden müssen, da noch andere Unsicherheiten bzw. Verteilungen in die Gesamtabschätzung einfließen. Es wurde diskutiert, ob z.B. ein 90. Perzentil aus (lokaler) Expositionsverteilung (PEC) hinsichtlich der Erreichung der Vorgabe zur Gesamtsicherheit ausreichend sei. Die Diskussion verlief ergebnisoffen.

- Die Unsicherheit in den Geodaten (Flächenutzung, Abstände, usw.) ist zu minimieren.
- Eine systematische Betrachtung aller Verfahrensunsicherheiten ist notwendig.
- Im statischen Ansatz werden nur Einträge georeferenziert (und probabilistisch) betrachtet, da das Gewässervolumen und somit die PEC nach Standardmodell berechnet werden.
- Im Fließgewässer bestimmt die georeferenzierte Tiefe das Volumen und somit die Konzentration.
- Die Dispersion im Fliessgewässer hat (in den bisherigen Berechnungen) relativ geringen Einfluss auf die PEC-Profile. Entscheidend sind die Behandlungswahrscheinlichkeit und das Behandlungszeitfenster. PEC-seitig entscheidet das Zeitfenster über das „Schutzniveau". Es sind weiterführende Modellbetrachtungen zu dem Fliessgewässeransatz erforderlich, die vorgestellten Ergebnisse waren erst der Anfang.
- Die wirkliche Verteilung der Windrichtungen spielt bei Nutzung der 90-Perzentile keine Rolle
- Die generische Hotspot-Ermittlung kann kulturspezifisch erfolgen, da RAC, Hecken, ufernahe Krautvegetation kulturspezifisch betrachtet werden können.
- Ein Refinement über Fate-Parameter dagegen ist substanzspezifisch.

Weiterhin wurde dann am 2. Workshoptag das Thema der **Hotspot Kriterien** (s. Präsentation U. Hommen in Anlage 5) behandelt. Das vom UBA 2007 vorgeschlagene Kriterium ist: Ein Hotspot, also eine ökologisch kritische räumliche Häufung hoher Exposition, liegt dann vor, wenn auf 1000 m Gewässerlänge in mehr als 10 % der Segmente die RAC durch die PEC überschritten wird oder wenn in einem Segment die PEC > 10 * RAC ist. Dieses Kriterium kann als protektiv angesehen werden, da konservativ bei einer RAC-Überschreitung in einem Gewässersegment immer direkt von 100 % Mortalität ausgegangen wird (also sehr steile Dosis-Wirkung und keine sublethalen Effekte).

Für eine generische Identifikation potentieller Hotspots in der Landschaft ist solch ein Kriterium angemessen; jedoch kann auf der Grundlage von Dosis-Wirkungsbeziehungen realer Substanzen die „Alles-oder-Nichts-Antwort" durch eine Dosis-Wirkungskurve mit einer „realistic-worst-case"-Steigung ersetzt werden. Es wurde eine Auswertung für Carbaryl vorgestellt (Slope = 4). Für eine mittelabhängige Risikoabschätzung (im Rahmen der Zulassung) kann die Steigung der Dosis-Wirkungskurve des für die RAC-Ableitung verwendeten Tests herangezogen werden. Weiterhin wurden für die Zulassung spezifische Hotspot Kriterien für verschiedene Organismengruppen abgeleitet, wozu Monitoringdaten, Traitdatenbanken, Fallstudien zur Wiedererholung und Populationsmodelle herangezogen werden. Für die realistischere Berücksichtigung von kurzen Expositionen, wie sie in Fließgewässern zu erwarten sind, wurde ein empirischer konservativer Ansatz vorgestellt.

B. Golla stellte anschließend das Konzept eines web-basierten Berechnungstools zur generischen und zur mittelspezifischen PEC-Berechnung und Hotspot-Analyse vor:

Von Seiten der Behörde würde demnach die Erstellung von Geofachdaten- und Expositionsdatenbank erfolgen, welche in gewissen Zyklen aktualisiert würden. Ein web-basiertes Tool erlaubt die generische Analyse der potentiellen Hotspots. Für die spätere Mittel-Zulassung erlaubt das Tool die Eingabe der substanzabhängigen Daten zu Anwendung und Toxizität. Ausgegeben werden die Anzahl der Risikosegmente und der daraus entstehenden Hotspots. Eine Verfeinerung der Geofachdaten innerhalb eines einzelnen Zulassungsverfahrens ist nicht vorgesehen. Dies kann nur im Rahmen der zyklischen Aktualisierung des Datenbestandes erfolgen und steht damit allen Antragstellern zur Verfügung.

Nach diesen einführenden Vorträgen wurden folgende Themen in zwei Sessions mit je drei Arbeitsgruppen diskutiert (Protokolle s. Anlage 3):

AG 1: Bewertungsmaßstäbe und Definition des Schutzniveaus (Toni Ratte + Michael Klein)

In der AG 1 bestand Einigkeit, dass der relevante Gewässerabschnitt im Zusammenhang mit der Ausdehnung der lokalen Population gesehen werden muss. Das UBA sieht als zeitliche Dimension, dass für das Zulassungsfenster (10 Jahre) mit genügend großer Sicherheit (95 %) gewährleistet sein muss, dass keine unvertretbaren Auswirkungen auf die lokale Population auftreten[3].

Letztendlich sei, nach Ansicht des UBA, nicht die Sicherheitsstufe des bisherigen Verfahrens die Richtschnur, sondern die Maßgabe, dass Populationen mit 95% Sicherheit zu schützen sind. Ein Vergleich mit der Sicherheit des bisherigen Verfahrens sei daher nicht notwendig und wäre auch wegen des Fehlens eines quantitativ fassbaren Schutzniveaus (z.B. 95%) nicht auf direktem Wege möglich. Die Teilnehmer der Gruppe sahen die eigentliche Schwierigkeit für das neue Verfahren weniger in der Schutzzieldefinition, sondern eher in der „Übersetzung“ der Modellierungsergebnisse in Vorhersagen, inwieweit das definierte Schutzziel mit den abgeleiteten Risikomanagementmaßnahmen erreicht werden kann.

Über das Schutzniveau seiner Fassung und seiner Interpretation bestand kein Konsens in der Diskussion. Auch das Schutzziel wurde kontrovers diskutiert, z.B. ob grundsätzlich alle lokalen Populationen im System geschützt werden sollen.

Der Trait-Ansatz sei grundsätzlich akzeptabel, wenn er alle Arten einschließe, die typischerweise in wenig belasteten Agrargewässern vorkommen.

AG 2: Bewertung unter regulatorischen + ökologischen Gesichtpunkten (Christoph Schäfers + Martina Roß-Nickoll)

Die Teilnehmer waren sich einig, dass nach einem erfolgten landschaftsbezogenen generischen Hotspot-Management der Übergang vom alten zum neuen System einfach sein sollte und votierten für die parallele Nutzung beider Systeme in der Übergangsphase. Die Finanzierung des Systems wurde als schwierig angesehen und die Verknüpfung mit anderen behördlichen Aufgaben und Landschaftsmaßnahmen als eine wichtige Option herausgearbeitet. Dies würde auch Widersprüche mit anderen Schutzzielen – z.B. aus dem Naturschutz verhindern helfen.

AG 3: Bewertung unter sozio-ökonomischen Gesichtspunkten (Martin Bach + Matthias Trapp)

Die Vorteile eines neuen Bewertungssystems (realistischescher Risikobewertung, , Vereinfachung und Erleichterung des substanzbezogenen Risikomanagements) wurden in der Diskussion anerkannt. Viele Diskussionsteilnehmer sahen die Umsetzung des Verfahrens jedoch als schwierig

[3] S. dazu auch Folie 2 in der Präsentation von J. Wogram zum Schutzziel: Die Vorgabe ist nach EU Dir 91/414 „Unter Bedingungen der Anwendung keine nachhaltigen Schäden an Populationen von Nichtzielorganismen“ und nach dem „Paraquat-Urteil ist das Sicherheitsmaß die „an Sicherheit grenzende Wahrscheinlichkeit“.

an, da u.U. erhebliche Kosten bei der Implementierung von landschaftsbezogenen Risikomanagementmaßnahmen zu erwarten sind. Zum jetzigen Zeitpunkt sahen weder die Industrie noch die landwirtschaftlichen Interessenvertreter oder der amtliche Dienst Möglichkeiten eines Engagements bei der Umsetzung der landschaftsbezogenen Risikomanagementmaßnahmen.

Die AG betonte, dass die Vorteile eines neuen Verfahrens noch besser aufgezeigt und dargestellt werden müssten. Auch eine Schärfung der Aussagen – z.B. ein „bestechend einfaches Verfahren“ könnte nutzen. Weiterhin votierten die Diskussionsteilnehmer für die Erprobung des Verfahrens in einem Pilotgebiet. Dazu sollte die Frage, wie viel Risikosegmente mit dem neuen Verfahren in den Sonderkulturen entstehen, möglichst rasch innerhalb des Projektes beantwortet werden.

AG 4: Bewertung des Risikomanagements unter regulatorischen, ökologischen und sozioökonomischen Gesichtspunkten (R. Kubiak / Udo Hommen)

Die Gruppe sprach sich zunächst erstmal eindeutig für den Einsatz des dynamischen Fließgewässermodells in einem georeferenzierten probabilistischen Ansatz aus.

Im Lauf der Diskussion wurde folgende Roadmap entwickelt:

1. Abschluss des GeoRisk Projekts
    - Sorgfältige Dokumentation des Ansatzes
    - Ergebnisse: bundesweit für Obst, Hopfen, Wein mit dem statischen Gewässermodell auf ATKIS-Basis und Standardgrabenszenario
    - Beispielrechnungen mit dem dynamischen Fließgewässermodell
2. Politische Entscheidung, ob GeoPRA und bundesweites substanzunabhängiges (d.h. generisches) Management der Hotspots gewünscht sind
3. GeoRisk Review durch externe Fachleute, Einigung auf Modellansatz / Annahmen / Parameter
4. Managementsegmentberechnung für Pilotgebiet (Hallertau)
5. Runder Tisch mit allen Beteiligten (Politik, Landwirtschaft, Behörden) zur Entscheidung über Pilotgebiet auf der Basis der Ergebnisse von 4
6. Einführung im Pilotgebiet Hallertau mit Monitoring (Sammeln von Erfahrungen in der praktischen Umsetzung des RMA-Managements, chemisches und biologisches Monitoring)
7. Einführung in weiteren Raumkulturen auf Basis der Erfahrungen im Pilotgebiet Hallertau (ggf. nach erforderlicher Modifikation der Methoden)

AG 5: Technische Implementierung des Verfahrens – Anforderung der Nutzer (B. Golla, J. Krumpe)

Die Arbeitsgruppe diskutierte sehr kontrovers. Es bestand grundsätzlich Konsens über die Grundzüge und Ziele der generischen Analyse und Bewertung von Hotspots sowie, dass dies entsprechende Werkzeuge zur Bearbeitung bedarf. Es bestand kein Konsens über die Notwendigkeit die zukünftige Zulassungsentscheidung von spezifischen PSM von einzelnen Hotspots abhängig zu machen und damit, grundsätzlich über die Notwendigkeit und den Einsatz des hier vorgestellen Tools. Desweiteren erzeugten die soweit vorliegenden Konzepte der zentralen Geodatenverwaltung zahlreiche Fragen, etwa nach den Kosten oder der Flexibilität und damit möglichen Widersprüchen zu vorgeschlagenen Zyklus der Verfeinerung im Zulassungsverfahren. Der IVA schlug eine Alternative für die zukünftige Gestaltung der Bewertung spezifischer PSM vor (siehe Anhang). Eine Demoversion wurde vorgestellt.

Auf der Basis einer vom JKI entwickelten Demoversion des WebTools wurden verschiedene Aspekte der Handhabung diskutiert. In diesem Zusammenhang wurden folgende Anforderungen an die technische Umsetzung formuliert:

- Transparenz des Berechnungsverfahrens
- Sicherheit und Vertraulichkeit übermittelter Daten
- Gute Zugänglichkeit der Geodaten und Werkzeuge. Nach Vorstellung des IVA sollten die Geodaten und Rechenmodelle den Firmen "off-line" zur Verfügung stehen
- Möglichkeit der Einbringung hochaufgelöster Geodaten im Zulassungsverfahren einzelner Mittel
- Verfeinerungsoptionen der Expositionsanalyse
- Regelmäßige Datenaktualisierung

AG 6: Übertragbarkeit auf Feldkulturen – Risikoabschätzung und Managementoptionen (Martin Bach und Michael Klein)

Die Notwendigkeit einer PRA Drift für Feldkulturen wurde als wesentlich geringer angesehen als für Raumkulturen. Unter anderem wurde das auch deshalb so eingeschätzt, weil hier die Drift gemessen an den Einträgen aus Runoff und Drainage nicht das zentrale Problem darstellt. Wegen der im Ackerbau üblichen Kulturfolgen und damit einhergehender unterschiedlicher Bewertungsgrundlagen wären die Unschärfen eines PRA Ansatzes signifikant größer als in den Sonderkulturen.

3. Tag

Am dritten Tag wurde nach einer Zusammenfassung der Kernaussagen des 2. Tages durch Roland Kubiak (s. Folien in Anhang 5) der vorgestellte Ansatz im Plenum anhand folgender Punkte intensiv diskutiert (die Beiträge finden sich im Anlage 4):

Schutzniveau für Oberflächengewässer (Einführung Jörn Wogram)

Die gesetzlichen Anforderungen besagen, dass die Anwendung von Pflanzenschutzmitteln keine nachhaltigen Schäden an Populationen von Nichtzielorganismen mit an Sicherheit grenzender Wahrscheinlichkeit“ auftreten sollen, wobei das Schutzgut Populationen, nicht Individuen sind. Als Konkretisierung von „mit an Sicherheit grenzende Wahrscheinlichkeit“ wird 95 % Sicherheit für ein Jahr über die Zulassungsdauer von 10 Jahren angesehen

Unsicherheiten des neuen Verfahrens (B. Golla / M. Trapp)

ATKIS bietet die zur Zeit bestmögliche flächendeckende Abbildung, der Gewässer, wobei die Unsicherheiten bis zum Ende des Vorhabens eingeschätzt werden müssen und in Zukunft mit

InVeKoS die Applikationsflächen bundesweit in HR zur Verfügung stehen. Weitere zu quantifizierende Unsicherheiten betreffen die hydrologischen Eingangsparameter (z.B. Breite/Tiefe Verhältnisse, Fließgeschw., Abflußmenge), die prinzipiell verfügbar sind, während Applikationsmuster (Behandlungswahrscheinlichkeit und -zeitpunkt) unsicher und variabel sind.

Managementmaßnahmen an potenziellen Überschreitungssegmenten (Martin Bach)

Prinzipiell stehen für ein landschaftsbezogenes substanzunabängiges Risikomanagement folgende Maßnahmen zur Verfügung: Rodung gewässernaher Reihen, Anpflanzen von Hecken (oder anderer driftmindernder Vegetation), Anlage von naturnahen Uferstreifen, Verschieben oder Unterlassen der Böschungsmahd und technische Maßnahmen (Driftschutzzäune). Auf der ökologischen Seite kann durch z.B. die Anlage von Regenerationsräumen das Wiederbesiedlungspotential verbessert und somit Hotspots entschärft werden. Bei der Planung von Maßnahmen sind Wirksamkeit und Kosten, Organisation und Durchführung, Synergien mit anderen Regelwerken sowie die Möglichkeiten der Erfolgskontrolle zu berücksichtigen.

Den Abschluss bildete das **Meinungsbild des Plenums** zum neuen Ansatz. Aus dieser Diskussion bleibt festzuhalten, dass der **IVA** (s. Folien von F. Dechet) grundsätzlich den georeferenzierten probabilistischen Ansatz mit einer generischen Hotspot Identifizierung und darauf aufbauendem Risikomanagement in der Landschaft unterstützt. Landschaftsbezogenes Risikomanagement und Zulassung sollen aber konzeptionell und verfahrenstechnisch getrennt sein; eine Zulassung sollte auf Basis einer bundesweiten Verteilung (und nicht einer Hotspot-Analyse) durchgeführt werden.

Vorgeschlagen wurde, über „geoEckwerte Aquatik" die Resultate einer Georeferenzierten probabilistischen Analyse als Weiterentwicklung der bisherigen Drifteckwerte zu verwenden. Für das weitere Vorgehen wünschte der IVA eine engere Einbindung von Landwirtschaft und Industrie und die Koordinierung aller Aktivitäten durch eine verantwortliche Stelle, z.B. durch BVL-Beirat, die dann einen Ablaufplan für die Implementierung inklusive Pilotstudie erstellen soll.

In der anschließenden Diskussion wurde deutlich, dass eine aktive koordinierende Stelle für die Implementierung eines georeferenzierten und probabilistischen Ansatzes von allen Teilnehmern gewünscht wird.

Aus Sicht des UBA wurde auf dem Workshop Konsens darüber erzielt, dass die vorgestellte dynamische Expositionsmodellierung sowie die trait basierte Effektbewertung notwendige und Erfolg versprechende Ansätze darstellen und dass die am Vortag entwickelte Roadmap zur Implementierung allgemein begrüßt wurde.

Anlage 1: Agenda

16.11.2009

Moderation: Roland Kubiak, Rapporteur: Christoph Schäfers

13:00 Begrüßung und Einführung in die Thematik (J. Wogram, R. Kubiak)

13:20 Die Ergebnisse des UBA Workshops von 2007 (S. Matezki)

13:35 Ein neues Konzept zur Beurteilung von Drifteinträgen aus Raumkulturen in Oberflächengewässer (U. Hommen)

14:05 Erste Verständnisfragen zum Gesamtansatz

14:15 Berechnung der Drifteinträge und PEC-Berechnung Standardgewässer (M. Klein)

14:35 Ergebnisse der Abschätzung von Hotspots und Managementsegmenten auf der Basis des Standardgewässermodells (B. Golla)

14:55 Unsicherheiten der Geodaten (J. Krumpe, M. Trapp)

15:15 Diskussion der PEC Berechnungen mit Standardgewässerszenario: Annahmen und Unsicherheiten

15:45 Kaffeepause

16:15 Die Einbeziehung von Fließgewässern in das Verfahren – Grundlagen und Datenanforderungen (M. Bach)

16:40 Die Einbeziehung von Fließgewässern in das Verfahren – Erste Umsetzung (M. Trapp)

17:00 Diskussion Fließgewässeransatz

17:30 Ende des fachlichen Teils am 1. Tag

18:30 Geführte Besichtigung des Bauhaus-Museums, Gropiusallee 38 mit anschließendem gemeinsamen Abendessen im Bauhaus Klub.

17.11.2009

Moderation: Roland Kubiak, Rapporteur: Martina Roß-Nickoll

08:30 Zusammenfassung der Kernaussagen des 1. Tages (R. Kubiak)

08:45 Ableitung von Hotspot Kriterien (U. Hommen)

09:15 Diskussion

09:40 Vorschlag zur technischen Umsetzung des geplanten Zulassungsverfahrens (Burkhard Golla, Jens Krumpe)

10:00 Diskussion

10:15 Einführung in AP 4 und Aufteilung des Plenums in Arbeitsgruppen (R. Kubiak)

10:35 Kaffeepause

11:00 Arbeitsgruppen zur Bewertung des Verfahrens zur Risikoabschätzung

1. Bewertungsmaßstäbe und Definition des Schutzniveaus (Toni Ratte + Michael Klein)
2. Bewertung unter regulatorischen + ökologischen Gesichtpunkten (Christoph Schäfers + Martina Roß-Nickoll)
3. Bewertung unter sozio-ökonomischen Gesichtspunkten (Martin Bach + Matthias Trapp)

13:00 Mittagspause

14:00 Plenum mit Ergebnissen der drei AGs

14:50 Fragestellungen und Aufteilung in Arbeitsgruppen

15:00 Kaffeepause

15:30 Arbeitsgruppen:

4. Bewertung des Risikomanagements unter regulatorischen, ökologischen und sozioökonomischen Gesichtspunkten (R. Kubiak / Udo Hommen)
5. Technische Implementierung des Verfahrens – Anforderung der Nutzer (B. Golla, J. Krumpe)
6. Übertragbarkeit auf Feldkulturen – Risikoabschätzung und Managementoptionen (M. Bach / M. Klein)

17:30 Plenum mit Ergebnissen der beiden AGs

18:15 Ende des fachlichen Teils am 2. Tag

19.00 (ab) Gemütliches Nachtreffen der Teilnehmer im Brauhaus „Alter Dessauer“

18.11. 2009

Moderation : Roland Kubiak, Rapporteur Udo Hommen

08:30 Zusammenfassung der Kernaussagen des 2. Tages (Roland Kubiak)

Plenumsdiskussion zu den Punkten:

09:00 Unsicherheiten des neuen Verfahrens

09:30 Schutzniveau für Oberflächengewässer

10:00 Kaffeepause

10:30 Managementmaßnahmen an potenziellen Überschreitungssegmenten

11:00 Meinungsbild des Plenums zum neuen Ansatz

11:45 Meinungsbild des Umweltbundesamtes

12:00 Ende des Workshops

Anlage 2: Workshopteilnehmer

GeoRisk-Konsortium:

Bach Martin (MB)

Golla Burkhard. (BG)

Hommen Udo (UH)

Klein Michael (MK)

Krumpe Jens (JK)

Kubiak Roland (RK)

Ratte Toni (TR)

Roß-Nickoll Martina (MR)

Schäfers Christoph (CS)

Trapp Matthias (MT)

Guerniche Djamal (DG)

Umweltbundesamt:

Matezki Steffen (SM)

Müller Alexandra (AM)

Osterwald Anne (AO)

Pickl Christina (CP)

Wogram Jörn (JW)

Beirat:

Dechet Friedrich (FD)

Glas Michael (MG)

Klein Manfred (MaK)

Liess Matthias (ML) (nur am 17.11.)

Morgenstern Michael (MM)

Schad Thorsten (TS)

Streloke Martin (MS)

Weitere:

Berger Gert (GB)

Erzgräber Beate (BE)

Fischer Reinhard (RF)

Spickermann Gregor (GS)

Kerber Martin (MKe)

Mair Jakob (JM)

Matties Michael (MiM)

Mendel-Kreusel Renate (RMK)

Rautmann Dirk (DR)

Resseler Herbert (HR)

Schriever Carola (CaS)

Anlage 3: Protokolle aus den Arbeitsgruppen

AG 1: Bewertungsmaßstäbe und Definition des Schutzniveaus

Teilnehmer/innen: Michael Morgenstern, Thorsten Schad, Carola Schriever, Dirk Rautmann, Gregor Spickermann, Michael Matthies, Steffen Matezki, Anne Osterwald

Leitung: Toni Ratte & Michael Klein

Zur Definition des Schutzziels bestand Einigkeit, dass der relevante Gewässerabschnitt im Zusammenhang mit der Ausdehnung der lokalen Population gesehen werden muss. Für die zeitliche Dimension muss dabei mit genügend großer Sicherheit gewährleistet sein, dass für das Zulassungsfenster (10 Jahre) keine unvertretbaren Auswirkungen auf die lokale Population auftreten. Dabei muss nicht berücksichtigt werden, dass sich innerhalb der 10 Jahre die Landschaft verändert/verändern könnte und so evtl. das das Schutzziel nicht mehr erreicht wird. Die Zulässigkeit von Landschaftsveränderungen wird durch andere Regelungen bestimmt.

Falls sich außerhalb der Pflanzenschutzmittelregelung grundsätzlich die Landschaft in dieser Zeit ändert, würde diese Situation über andere gesetzliche Regelwerke gesteuert werden können.

Für den Vergleich des bisherigen Verfahrens mit dem neuen Bewertungssystem ist es der Wunsch des UBA, dass alle unter dem bisherigen Verfahren zugelassenen Wirkstoffe ihre Zulassung behalten können. Der Eindruck vom bisherigen Verfahren war, dass es - obwohl nicht quantifizierbar - in der Praxis zufriedenstellend funktioniert hatte. Letztendlich wäre ja nicht die Sicherheitsstufe des bisherigen Verfahrens die Richtschnur, sondern die Maßgabe, dass Populationen mit 95% Sicherheit zu schützen sind. Ein Vergleich mit der Sicherheit des bisherigen Verfahrens ist daher nicht notwendig und wäre auch wegen des Fehlens eines quantitativ fassbaren Schutzniveaus (z.B. 95%) nicht auf direktem Wege möglich.

Einige Teilnehmer der Gruppe sahen die eigentliche Schwierigkeit für das neue Verfahren weniger in der Schutzzieldefinition, sondern eher in der „Übersetzung“ der Modellierungsergebnisse in Vorhersagen, inwieweit das definierte Schutzziel mit den abgeleiteten Risikomanagementmaßnahmen erreicht werden kann (z.B. Schutzziel mit Populationen als Bezugssystem und Bewertungsverfahren mit der Prognose für eine PEC-Wahrscheinlichkeit an einzelnen Segmenten als Bezugssystem)..

Es bestand keine Einigkeit darin, ob grundsätzlich alle lokalen Populationen im System geschützt werden sollen. Schwierigkeiten werden hier darin gesehen, wie in Gewässersystemen lokale Populationen abzugrenzen sind. Wenn ein 10m Segment für eine lokale Population (etwa von Insektenlarven) steht, wäre jedes Segment zu schützen, was erhebliche Auswirkungen auf die Landwirtschaft hätte.

Der Trait-Ansatz ist grundsätzlich akzeptabel, wenn er alle Arten einschließt, die typischerweise in wenig belasteten Agrargewässern vorkommen.

Von Teilnehmern wurde diskutiert, das neue System als eine Kombination von georeferenzierter Analyse und Szenario-basierten Ansätzen zu realisieren. Bei diesem Verfahren würde zunächst eine generische Analyse durchgeführt werden, um die Hotspots zu identifizieren und zu regeln. Anschließend würde eine substanzspezifische Analyse basierend auf einem realistic worst case Szenario durchgeführt werden, das für alle Wirkstoffe mit Hilfe der generischen Analyse im ersten

Schritt definiert werden müsste. Der Unterschied zu dem aktuell vorgeschlagenen Ansatz wäre, dass bei diesem Verfahren eine repräsentative Gesamtheit von Gewässerabschnitten berücksichtigt werden würde, während im vorgeschlagenen Verfahren bei der substanzspezifischen Analyse das Auftreten einzelner Hotspots über die Zulassung eines Wirkstoff entscheiden würde.

Die AG sprach sich für die Berücksichtigung von Fließgewässern im Bewertungsverfahren aus, weil sie in Deutschland eher die Regel als die Ausnahme sind. Es wurde vorgeschlagen, ein Metamodell zur Berechnung in Erwägung zu ziehen, welches auf dem kompletten Fließgewässersystem basiert. Man würde so die wesentlichen Ergebnisse reproduzieren können, ohne das Fließgewässermodell jedes Mal neu durchrechnen zu lassen. Voraussetzung wäre jedoch, dass das originale (dynamische) Fließgewässermodell wenigstens einmal komplett für alle relevanten Gewässer durchgerechnet würde.

Die AG diskutierte ausführlich die verschiedenen Unsicherheiten der Bewertungsverfahren auf der Fate-Seite. Der Eindruck war, dass es durch den georeferenzierten Ansatz erstmals möglich würde, die tatsächlichen Konzentrationen vor Ort (gemessen) mit Werten aus den konservativen Prognosemodellen zu vergleichen und zu überprüfen.

Weitere Anmerkungen der Teilnehmer zum Thema Unsicherheiten:

- Auf eine korrekte Beschreibung von Unsicherheiten (aufgrund Messungenauigkeiten) und Variabilitäten (aufgrund natürlicher Streuungen) sollte geachtet werden. Unsicherheiten mancher Parameter (z.B. Driftrate) sollte man daher besser als natürliche Variabilitäten bezeichnen, da diese nicht durch zusätzliche Messungen reduziert werden können. In jedem Fall sollte es das Ziel sein, die das Ergebnis bestimmenden Variabilitäten in das Modell adäquat einzubauen und zu berücksichtigen.
- Wenn ein Parameter sehr variabel und sensitiv ist, sollte er in die Monte-Carlo-Simulation eingehen. (Beispiel: Variation der Tiefe über mehrere Jahre oder auch über eine Saison). Dabei könnten mehrdimensionale Monte-Carlo-Analysen zur quantitativen Beschreibung der Untersicherheiten helfen. Bei derartigen Analysen wären aber zusätzliche Informationen zur Abhängigkeitsfunktion notwendig, um unrealistische Parameterkombinationen zu vermeiden.
- Grundsätzlich wäre ein Ranking der Parameter hinsichtlich Sensitivität und Variabilität/ Unsicherheit nützlich.

Neben diesen allgemeinen Hinweisen wurden auch Anmerkungen zu Unsicherheiten einzelner Parameter gemacht:

- Unsicherheiten bei Messung der Georeferenzen (Gewässerabschnitte, Applikationsflächen, Auflösung) könnten als Prozent nicht erfasste Abschnitte bzw. Flächen berücksichtigt werden.
- Unsicherheit durch Variabilität bei Messung der Windrichtungen (nicht georeferenziert) könnten in Form von Perzentilen probabilistisch berücksichtigt werden.
- Unsicherheiten (durch Variabilität) der Driftminderungsfaktoren (Applikationstechnik) könnten in Form von Prozentwerten deterministisch berücksichtigt werden.
- Unsicherheit der Driftminderungsfaktoren in der Landschaft (räumlich und zeitlich variabel) könnten ebenfalls in Form von Prozentwerten deterministisch berücksichtigt werden.

- Unsicherheiten durch Variabilität von Deposition, Dispersion der Substanz im Gewässer (incl. Expositionszeitfenster) könnten durch Perzentilen probabilistisch berücksichtigt werden.
- Weitere Faktoren, die derzeit nicht in die PEC-Berechnung eingehen (Abschirmung), sollten für eine Bewertung der Konservativität des Modells herangezogen werden.
- Hinweis: Düsenreduktionsfaktoren sind konservativ: 75 % Driftreduktion bedeutet in der Realität zwischen 75 % und 90 %

Die Arbeitsgruppe diskutierte auch Unsicherheiten, die sich aus der Bewertung der Effekte ergeben. Es wurden folgende Vorschläge gemacht:

- Monitoring durchführen, um Unsicherheiten über die Erreichung des Schutzziels zu reduzieren
- stärker vorhandene Monitoring-Untersuchungen berücksichtigen;
- überprüfen, ob Traits richtig zugeordnet sind;
- untersuchen, wie man von Standardorganismen auf empfindliche Arten schließen kann.

Die Arbeitsgruppe stellte fest, dass das derzeitige Bewertungssystem keine Georeferenzierbarkeit bzgl. der Effekte berücksichtigt: Es spielt deshalb keine Rolle, ob identifizierte empfindlichste Arten überall vorkommen oder ob das Potential für eine Wiederbesiedlung überhaupt existiert. In der Diskussion wurde festgestellt, dass die Zusammensetzung der Gemeinschaften grundsätzlich ebenfalls georeferenzierbar wäre, dies aber derzeit nicht vorgesehen ist.

AG 2: Risikobewertung unter regulatorischen + ökologischen Gesichtpunkten

Teilnehmer/-innen: Gert Berger, Friedrich Dechet, Reinhard Fischer, Burkhard Golla, Manfred Klein, Renate Mendel-Kreusel, Michael Morgenstern, Herbert Resseler, Martin Streloke, Jörn Wogram

Leitung: Christoph Schäfers & Martina Roß-Nickoll

A. Bewertung der vorgeschlagenen (Übergangs-) lösungen bis zur effizienten Umsetzung von landschaftsbezogenen „Hotspot"-Maßnahmen unter regulatorischen Gesichtpunkten (z.B. Schutzzielerreichung)

Es herrschte Einigkeit darüber, dass ein landschaftsbezogenes Hotspots-Management nur generisch erfolgen kann. Änderungen der einmal erfolgten Klassifikation der Hotspots wären nur bei Änderungen der Nutzung vermittelbar. Dass die Hotspotdefinition über längere Zeiträume nicht statisch bleiben kann, wurde schon durch zu erwartende Nutzungsänderungen als vorgegeben gesehen. Solche Nutzungsänderungen sind gerade im Weinbau aufgrund der gesetzlichen Voraussetzungen zu erwarten.

Der Übergang vom alten zum neuen System sollte einfach bleiben; eine Umsetzung ist erst nach Abschluss des generischen Hotspot-Managements denkbar, bis zu diesem Zeitpunkt sollte das alte System weiterhin genutzt werden. Auch wenn vermutet wurde, dass sich der Übergang als in

Anbetracht des Regelungsbedarfs der Zuständigkeiten und der Finanzierung langwierig gestalten könnte, wurde dringend empfohlen auch diese Übergangsphase so einfach wie möglich zu gestalten.

Hinsichtlich der Finanzierungsmöglichkeiten des Hotspot-Managements wurden verschiedene Aspekte diskutiert. Im Moment gibt es aus Sicht der Landwirte wenig Anreize, die es rechfertigen, in Maßnahmen zu investieren. Da es sich für einen einzelnen Landwirt nicht lohnt etwas für die Allgemeinheit zu finanzieren, wurde über die Einbeziehung von Nachbarschaftsbeziehungen in kleineren Konsortien nachgedacht.
Im Obstbau, Wein und Hopfen gibt es grundsätzlich eine Bereitschaft zur Umsetzung, da einige Produkte wegfallen würden (pro Firma 3-7), wenn die 20 m-Auflage ohne HS-Management entfallen würde.

Es wurde auch die Möglichkeit einer Umsetzung auf der kommunalen Ebene (untere Landschaftsbehörden, Wasserverbände, Landschaftsverbände) diskutiert. Weiterhin wurde die Einbeziehung in Maßnahmen der Flurbereinigungsverfahren oder die Umsetzung im Rahmen von Eingriffs-/Ausgleichsregelungen angesprochen.

B. Bewertung möglicher langfristiger Auswirkungen einer bundesweiten Umsetzung des landschafts- bezogenen Risikomanagements auf die biologische Vielfalt terrestrischer Ökosysteme (z.B. durch Erhöhung der Strukturvielfalt der Landschaft)

Die Gruppe war der Ansicht, dass auch wenn die avisierten Maßnahmen ein anderes Ziel haben, sie nicht im Widerspruch zu Landschaftsentwicklungs- und Naturschutzzielen stehen sollen. Es sollte kein anderes Schutzziel konterkarriert werden. Bevor erhebliche Landschaftsveränderungen vorgenommen werden, sollte immer erst geprüft werden, ob nicht auch ein Anwendungsmanagement sinnvoll möglich ist. Eine Unterstützung der Umsetzung im Rahmen von EU Umweltmaßnahmen (Agrarumweltmaßnahmen) wäre nur dann möglich, wenn keine Einbindung (Verpflichtung durch) in nationales Recht vorliegt.

Um die Neuanlage von ökologisch wertvollen Hecken zu fördern, werden nach bestehendem Recht keine Abstandsauflagen zu diesen neu angelegten Strukturelementen angewendet. Maßnahmen wie Trockenlegung und Verrohrung von Gewässern wurden aufgrund der vielfältigen ökologischen Nachteilswirkungen als nicht sinnvoll eingestuft. Bevorzugt sollten deshalb Maßnahmen angewendet werden, , die die Emission verhindern. Hecken, vor allem solche, die längere Zeit im Jahr unbelaubt sind, werden in der Diskussion insgesamt überbewertet. Bedeutsam ist auch die Einrichtung von Filterstreifen 20m (15), die nicht gespritzt werden.

Ein Hotspot Management würde eine hohe Sicherheit der Einhaltung gewährleisten, fände aber nur dann Akzeptanz, wenn die Zahl der zu managenden Hotspots nicht erheblich würde.

Die Durchführung einer Kosten-Nutzen-Analyse von Auflagen (20 m alt) und Maßnahmen (HS-Management) wurde für sinnvoll erachtet.

C) Bewertung der Kohärenz des neuen Bewertungs- und Managementansatzes mit anderen gesetzlichen Regelungsbereichen (z.B. mit Vorgaben zum Gewässerschutz)

Das neue System wird nicht im Widerspruch zur Wasserrahmenrichlinie (WWRL) sondern als Ergänzung dieser im Kleingewässerbereich gesehen. Das Monitoring im Rahmen der WWRL konzentriert sich schwerpunktmäßig auf größere Gewässer, in denen PSM meist fern der Eintragsorte mit hohen Verdünnungen zu verzeichnen sind.

Für den Fließgewässeransatz wären Überlegungen zum Effekt niedriger Konzentrationen bei längeren Expositionszeiten flussab nötig, was eine Trennung Risiko- Strecke und Management – Stecke bedeuten könnte.

Im Ackerbau können Stilllegungsstreifen am Rand kleiner Gewässer innerhalb von Ackerflächen die Ziele der FFH-Richtlinie (NATIRA2000) unterstützen (Schaffung von Lebensraum für Amphibienarten, wie Rotbauchunke oder Kammmolch) .

Hinsichtlich der Bewertung des Bewertungssystems soll Zuwanderung und Wiedererholung (Randgewässer) berücksichtigt werden. Die könnte dazu führen, dass es zu Verlusten von bis zu 10% z.B. der Fische auf Populationsebene kommen kann. Mögliche Kollisionen mit dem Tierschutz wurden erörtert.

AG 3: Bewertung unter sozio-ökonomischen Gesichtspunkten

Leitung: Martin Bach und Matthias Trapp

Teilnehmer/-innen: F. Dechet, B. Erzgraeber, M. Glas, M. Kerber, A. Müller, C. Pickl,

Als Junktim ist festzuhalten: kein neues Bewertungs- bzw. Zulassungsverfahren *ohne* Risikomanagementmaßnahmen.

Der wesentliche Nutzen eines geoPRA-basierten Zulassungsverfahrens wird in drei Punkten gesehen:

- Vereinheitlichung (Stichwort wurde ohne nähere Ausführungen dazu genannt, was im Einzelnen unter einer „Vereinheitlichung“ zu verstehen ist)
- Vereinfachung der bestehenden Anwendungsbestimmungen zum Schutz von Oberflächengewässern für den Eintragspfad Abdrift (bessere Verständlichkeit, Nachvolziehbarkeit undmehr Transparenz sollen die Praktikabilität und die Akzeptanz zur Einhaltung der Mindestabstände zu Gewässern erhöhen)
- Erleichterung bei der praktischen Anwendung von PSM durch Reduzierung der maximalen Mindestabstände (< 20m) aufgrund der realitätsnäheren Expositionsbewertung und RMA-Management.

Zum gegenwärtigen Zeitpunkt fühlt sich jedoch keine Institution berufen, (generische) Risikomanagementmaßnahmen in Sonderkulturgebieten zu etablieren bzw. eine entsprechende Initiative auf den Weg zu bringen.

- Für PSM-Hersteller ist ein Umsatzzuwachs durch ein Mehr an Behandlungsfläche infolge (möglicherweise) verringerter Abstandsauflagen unbedeutend.
- Die landwirtschaftliche Interessenvertretung (DBV) hat nach Meinung der TeilnehmerInnen bislang kein Interesse an der Neugestaltung der Anwendungsbestimmungen unter Berücksichtigung realitätsnäherer Risikobewertung und einem RMA-Management bekundet.
- Vertreter der PS-Dienste sehen die Bereitschaft der Länder, sich personell bzw. finanziell in diesem Bereich zu engagieren, äußerst skeptisch.

Die Vorteile eines neuen Verfahrens müssen den (potenziellen) Nutznießern somit offensichtlich noch besser aufgezeigt und nahegebracht werden; ein „bestechend einfaches“ Verfahren würde Vorteilhaftigkeit besser kommunizierbar machen

Für die weiteren Planungen zu Risikomanagement in Sonderkulturgebieten sind die relevanten Akteure an einen Tisch zusammen zu bringen, was eine wichtige Aufgabe des UBA-Projekts darstellen könnte. Kenntnisse der Überlegungen des BMELV zur Einbindung von Risikomanagementmaßnahmen in die rechtlichen Grundlagen (z.B. in das Zulassungsverfahren oder in den Nationalen Aktionsplan) sowie zu deren Durchführbarkeit und Finanzierbarkeit wären in diesem Zusammenhang sehr hilfreich. Synergien mit anderen Programmen wie bspw. Umsetzung WRRL, Flurneuordnung, Agrar(umwelt)programme, Eingriffs-Ausgleichsmaßnahmen u.a.m. sind aufzuzeigen und zu nutzen. Allerdings besteht das grundsätzliche Problem: wenn Risikomanagementmaßnahmen Bestandteil des Zulassungsverfahrens werden, dann sind diese Maßnahmen nicht aus EU-Mitteln förderfähig.

Eine Kernfrage im Zusammenhang mit der Konzeption und Umsetzung sollte vom Projekt möglichst bald beantwortet werden: Wie viele Kilometer Risikosegmente bleiben in Obst, Wein und Hopfen in DE tatsächlich übrig bei einem neuen Verfahren geoPRA (unter Einbindung Ansatz $PEC_{dynamisch}$ für Fließgewässer). Beispiel Hopfen (Hallertau): sind es eher 140 km oder 14 km?

Einführung und Umsetzung von Risikomanagementmaßnahmen sollten in einem Pilotprojekt exemplarisch erprobt werden. Aufgrund der guten Geodatenlage sowie der weiteren Vorarbeiten in der Region bietet sich dafür die Hallertau an; der Hopfenanbauer-Verband hat grundsätzlich Bereitschaft zur Zusammenarbeit bekundet.

Für die Einführung und Umsetzung von räumlich verorteten Maßnahmen können weiterhin Erfahrungen aus ähnlichen Bereichen nutzbar gemacht werden:

- Kontrolle von Cross Compliance-Auflagen im Rahmen des InVeKoS-Systems
- Umsetzung der „schützenswerten Kleinstrukturen“ (auf Gemeindeebene in DE): Kleinstrukturenverzeichnis
- (Negative) Erfahrungen aus Baden-Württemberg (Obstanbau Bodenseegebiet) verdeutlichen, dass andere Fachverwaltungen (Wasserwirtschaft; Naturschutz) sehr frühzeitig in Konzeption eingebunden werden müssen, um Bereitschaft zur Mitwirkung zu erreichen.

Zu einzelnen Maßnahmen:

- Unterlassung der Behandlung der gewässernächsten Reihe wird nicht als zielführend eingeschätzt: Einwanderung von Schadorganismen aus diesem Bereich in die Kernfläche und infolge dessen erhöhter Behandlungsaufwand insgesamt sind zu befürchten.
- Erfahrungen mit einer Rodung der letzten Reihe(n) liegen nur aus dem Alten Land vor. Dort wurde die Bereitschaft auf Seiten der Landwirtschaft zur Durchführung dieser Maßnahme jedoch vor allem durch die Alternative „massiv einschränkende Abstandsauflagen“ merklich befördert.
- Bei Anpflanzung von Hecken (auf dem Grund des Landwirts) wird möglicherweise die Gefahr gesehen, dass daraus nach einigen Jahren ein schützenswertes Landschaftselement entsteht, das nicht mehr gerodet werden darf, so dass diese Fläche dauerhaft einer landwirtschaftlichen Nutzung entzogen wäre. (Dieses Problem tritt allerdings auch bei anderen Agrarumweltprogrammen auf und ist dort durch einen Sonderregelung für derartige Landschaftselemente entschärft worden).

AG 4: Bewertung des Risikomanagements unter regulatorischen, ökologischen und sozioökonomischen Gesichtspunkten

Teilnehmer/-innen.	Friedrich Dechet, Michael Glas, Manfred Klein, , Toni Ratte, Herbert Resseler, Martina Roß-Nickoll Christoph Schaefers, Martin Streloke, Jörn Wogram
Leitung	Roland Kubiak & Udo Hommen

A. Allgemeine Diskussion zu GeoRisk und Ausblick

Es bestand Konsens darüber, dass in einem georeferenzierten probabilistischen Ansatz Fließgewässer dynamisch modelliert werden sollten.

In Bezug auf ein mögliches Vorgehen nach GeoRisk wurde diskutiert, ob Hopfen als Pilotkultur oder die Hallertau als Pilotgebiet besser geeignet wären. Eine Pilotkultur wäre als Einführung des Verfahrens für eine Kultur anzusehen und hätte daher ein stärkeres Gewicht als ein Pilotgebiet. Allerdings erscheint im Moment der Schritt zu einem Pilotgebiet Hallertau als besser umsetzbar und für alle Beteiligten als akzeptabel.

Zunächst ist eine politische Entscheidung und Abstimmung ausstehend, ob ein bundesweites Management (generisches Management in der Landschaft) gewünscht ist. Sobald das BMELV Bereitschaft signalisiert, könnte das Verfahren für Pilotgebiet eingeführt werden.

Dazu muss das Pilotgebiet komplett (dynamisch) durchgerechnet sein, um verbindliche Aussagen zur Zahl der Managementsegmente vorlegen zu können. Benötigt wird dazu von Matthias Trapp und Martin Bach eine grobe Einschätzung des Aufwandes für eine Risikoabschätzung für die gesamte Hallertau (z.B. Schließen von HR-Lücken, Fließgewässermodellierung).

Dann müssten alle Betroffenen (Politik, Landwirtschaft, Behörden, Industrie, Wissenschaft) an einen Tisch.

Es wurde darauf hingewiesen, dass die Unterschiede von ca. 140 km Managementsegmente in Hopfen nach AP2 (statisches Modell auf ATKIS-Basis mit Standardgrabenszenario) im Vergleich zu 11 km nach GeoPERA mit HR nicht nur durch HR, sondern auch durch unterschiedliche Annahmen (Verwendung des Medians statt des 90. Perzentils der lokalen PEC-Verteilung) zu Stande kommen.

AP2 errechnete ca. 140 km generisch plus weitere ca. 15 km spezifisch für den Beispielwirkstoff zu managende Gewässersegmente in der Kultur Hopfen bundesweit aus. Die Industrie wies darauf hin, dass aus ihrer Sicht nur ein generisches Risikomanagement, getrennt vom Zulassungsverfahren für einzelne Produkte, umgesetzt werden kann. Ansonsten könnten mit neuen Zulassungen unterschiedliche, zusätzliche Risikomanagementsegmente entstehen - ein Umstand, der in der Praxis kaum zu regeln sei. In diesem Zusammenhang wurde seitens Beratung und Industrie der Wunsch geäußert, **vor** der Diskussion des Managementbedarfs mit der Landwirtschaft in einem Pilotgebiet einen genaueren Überblick über die endgültige Größenordnung für diese Maßnahmen erhalten zu können (sind z.B. für 70, 20 oder 10 km Maßnahmen erforderlich?).

Es wurde der Wunsch nach einem 2. Pilotgebiet geäußert (Obstbau). Es ist zu prüfen, ob aus Berechnungen für die Hallertau auf der Basis von ATKIS und HR (aber gleichem Modell) z.B. eine Art „Korrekturfaktor" für die Anzahl der Managementsegmente nach ATKIS abgeleitet werden kann. Da die Unterschiede ATKIS zu HR aber stark von der Landschaftsstruktur bzw. der Intensität der landwirtschaftlichen Nutzung abhängen, müsste das Obstbaupilotgebiet ähnlich intensiv wie die Hallertau genutzt sein.

Es wurde auf die Notwendigkeit eines sorgfältig geplanten Monitorings in einem Pilotgebiet hingewiesen (chemisch, biologisch).

Zusammenfassend wurde die folgende **Roadmap** entwickelt:

1. Abschluss des GeoRisk Projekts
    - Sorgfältige Dokumentation des Ansatzes
    - Ergebnisse: bundesweit für Obst, Hopfen, Wein mit dem statischen Gewässermodell auf ATKIS-Basis und Standardgrabenszenario
    - Beispielrechnungen mit dem dynamischen Fließgewässermodell
2. Politische Entscheidung, ob GeoPRA und bundesweites substanzunabhängiges (d.h. generisches) Management der Hotspots gewünscht sind
3. GeoRisk Review durch externe Fachleute, Einigung auf Modellansatz / Annahmen / Parameter
4. Managementsegmentberechnung für Pilotgebiet (Hallertau)
5. Runder Tisch mit allen Beteiligten (Politik, Landwirtschaft, Behörden) zur Entscheidung über Pilotgebiet auf der Basis der Ergebnisse von 4
6. Einführung im Pilotgebiet Hallertau mit Monitoring (Sammeln von Erfahrungen in der praktischen Umsetzung des RMA-Managements, chemisches und biologisches Monitoring)
7. Einführung in weiteren Raumkulturen auf Basis der Erfahrungen im Pilotgebiet Hallertau (ggf. nach erforderlicher Modifikation der Methoden)

B. Managementoptionen

Die im Abschlussbericht zu Vorgängerprojekt (Schulz et al. 2007) enthaltenen Tabellen zu möglichen Optionen des Risikomanagements (Tab. 6.2, S. 81, s. Anhang) wurden kurz diskutiert.

Insbesondere wurde auf den Punkt „Verbreiterung der Uferstreifen" eingegangen. Es wurde diskutiert, ob ein verbreiterter Uferstreifen größer als die angestrebte maximale Abstandsauflage sein kann. Ziel des in GeoRisk vorgeschlagenen Verfahrens ist es, bei Beibehaltung des jetzigen Schutzniveaus die maximale Abstandsauflage für ein PSM von 20 auf 10 m zu verringern. Von Seiten der Industrie wurde hier aber der dringende Wunsch geäußert, dass eine 20 m Abstandsauflage auch weiterhin möglich sein sollte. J. Wogram gab zu bedenken, dass Vertreter der Pflanzenschutzämter der Länder wiederholt darauf hingewiesen haben, dass eine 20m-Auflage in Raumkulturen wirtschaftlich nicht vertretbar sei. Das UBA strebe daher eine Abschaffung dieser als nicht praxisgerecht bewerteten Auflage an. Über diesen Punkt der Beibehaltung einer Abstandsauflage von 20 m wurde kein Konsens erreicht.

Andere in der Gruppe vertraten die Auffassung, dass als ultima ratio (wenn in einzlnen Hotspots keine andere Möglichkeit der Eintragsreduzierung durch substanzunabhängiges Management möglich ist) auch Verbreiterung des Uferstreifens auf > 10 m eine Option sein könnte. Dabei wie auch bei den anderen Managementoptionen stellt sich die Frage nach der Akzeptanz/Durchsetzbarkeit einer solchen Maßnahme sowie ggfs. dem Ausgleich für den jeweisl betroffenen Landwirt (bzw. Flächeneigentümer). Eine abschließende Bewertung dieser Frage war in der Diskussion nicht möglich, da der zu erwartende Umfang des RMA-Managements noch nicht bekannt war.

Es wurde aber darauf hingewiesen, dass zumindest in der Übergangsphase (Einführung des Verfahrens bis zum abgeschlossenen Management der Hotspots) 20 m Auflagen weiterhin möglich sein sollen.

Die weiteren in der angehängten Tabelle gelisteten Managementoptionen wurden nicht näher diskutiert.

Als Alternative zur Anlage von (naturnahen) Hecken wurden das Anpflanzen von bestimmten Energiepflanzen (z.B. *Miscanthus*) als zu prüfende Alternative erwähnt.

Als weitere Option, auch gerade für die Übergangsphase,wurde der Einsatz von Driftnetzen / Driftzäunen in der Applikationsperiode genannt.

Tab. 6.2, S. 81 aus Schulz et al. 2007 (UBA-Bericht)

Hot-Spot-Maßnahmen

Tabelle 6.2: Übersicht über die Maßnahmen

	Zeit-rahmen	Effizienz	Umsetz-barkeit	Kosten	Akzeptanz	Kontrol-lierbarkeit	Verant-wortlich-keit
Landschaftsbezogene Maßnahmen							
Verbreiterung der Uferstreifen	schnell	mittel	schwierig		hoch	leicht	offen
	- konkret: Erhöhung des Abstands zwischen Kultur und Gewässer - umfasst keine spezielle Bepflanzung, Wirkung allein durch den Abstand						
Optimierung der Bepflanzung der Uferstreifen		hoch	schwierig		hoch	leicht	offen
	- gezielte Anpflanzung von driftreduzierenden Arten (schnell wachsend, dicht) - Eingriff in die lokale Situation (Beschattung) - ehemalige landwirtschaftliche Flächen können später kein Schutzgut werden (keine Probleme mit terrestrischen Schutzzielen) - Vorschrift (Auflagen im Alten Land): Randvegetation muss höher sein als die Kultur						
Optimierung der Pflege der Ufervegetation	schnell	hoch	leicht	gering	?		
	- z.B. Grasmahd-Zeitpunkt relativ zu Applikationszeiten						
Optimierung der Reihenanordnung		mittel	leicht	keine	hoch	leicht	Landwirt
	- Problem: Erosion und Runoff - evtl. eine Möglichkeit im Obstbau in Norddeutschland (im Rahmen der regelmäßig notwendigen Neuanpflanzungen)						
Umstrukturierung von Regenrückhaltebecken							
	- speziell im Weinbaugebiet - Problem: Definition oberhalb liegender Gewässerabschnitte als „Zuleitungen mit höherer Belastung" → Diskussion: Schutzgut - Möglichkeiten zur Reinigung in artificial wetlands						
Verbesserung des Potentials zur Wiederbesiedlung und -erholung							
Gewässerumbau am Hot Spot							
	- zur Verbesserung der Wiedererholung durch Erhöhung der Strukturvielfalt des Gewässers am Hot Spot - geringer Flächenbedarf - verringert die Fließgeschwindigkeit → Problem bei Gräben, die entwässern sollen (Obstbau: Wurzelfäule)						
Gewässerneuanlage		unklar	schwierig		?	leicht	offen
	- Schaffung von Refugien für Wiederbesiedlung durch Gewässerneuanlage - künstliche Gräbenstiche z.B. in angrenzenden Waldgebieten						
Anwendungsbezogene Maßnahmen							
Technische Driftminimierung		hoch	leicht	klein	hoch	gering	
Abstandsauflagen		hoch	leicht	klein	gering	gering	
Anwendung nur bei Windstille bzw. Wind vom Gewässer weg		hoch	schwierig	gering	gering	kaum	
Anwendungsverbot in Hot-Spot-Abschnitten		hoch	leicht	gering	gering	kaum	

AG 5: Technische Implementierung des Verfahrens – Anforderung der Nutzer

Teilnehmer/innen: A.Müller, M.Trapp, T.Schad, R.Fischer, G.Spickermann, B.Erzgräber, R.Mendel-Kreusel, D. Guemiche

Leitung: B.Golla, J.Krumpe

In der Arbeitsgruppe standen die folgenden Diskussionspunkte auf der Agenda:

- Nutzergruppe des Verfahrens
- Transparenz im Berechnungsverfahren
- Zugänglichkeit der Geodaten, Werkzeuge, Methode
- Sicherheit/Vertraulichkeit übermittelter Daten
- Sicherheit der Bewertung, Gültigkeit/Dauerhaftigkeit

Demo des Web-basierten Werkzeuges zur mittelspezifischen Ermittlung der Risikominderungsgruppe:

- von Antragstellern eingegebene Substanzdaten werden nicht zentral gespeichert
- Simulationstool (bereitgestellt über WEB-Interface und SOAP Schnittstelle) mit Beispielen aus der Landschaft (Transparenz in der Berechnung) macht Einfluss der Driftminderung, Perzentilwahl auf Loading/Konzentrationsberechnung deutlichPraktikabilität eines georeferenzierten Ansatzes bei der Zulassung von PSM wurde grundsätzlich diskutiert

Die Arbeitsgruppe diskutierte sehr kontrovers. Es bestand grundsätzlich Konsens über die Grundzüge und Ziele der generischen Analyse und Bewertung von Hotspots sowie, dass dies entsprechende Werkzeuge zur Bearbeitung bedarf. Es bestand kein Konsens über die Notwendigkeit die zukünftige Zulassungsentscheidung von spezifischen PSM von einzelnen Hotspot abhängig zu machen und damit, grundsätzlich über die Notwendigkeit und den Einsatz des hier vorgestellen Tools. Desweiteren erzeugten die soweit vorliegenden Konzepte der zentralen Geodatenverwaltung zahlreiche Fragen, etwa nach den Kosten oder der Flexibilität und damit möglichen Widersprüchen zu vorgeschlagenen Zyklus der Verfeinerung im Zulassungsverfahren. Der IVA schlug eine Alternative für die zukünftige Gestaltung der Bewertung spezifischer PSM vor (siehe Anhang)

Eine zentrale Datenhaltung, von bewertungsrelevanten Geoinformationen für das Zulassungsverfahren wird als grundsätzlich sinvoll diskutiert. Diese sollten nach einer abgestimmten Guideline für GeoPRA-Zwecke aufgearbeitet sein. Solche Guidance könnte in Gremien wie dem BVL-Beirat abgestimmt werden. Dies wird unabhängig von der Verwendung solcher Daten für die PSM-spezifische Hotspot-Analyse gesehen. Letztere wird seitens des IVA als nicht zielführend bewertet.

Mögliche Antragsteller wünschten verschiedene Verfeinerungsoptionen im Web-Tool:

- Einbringen hochaufgelöster Geodaten im Zulassungsverfahren einzelner Mittel
- Verfeinerungsoptionen für Hotspot-Kriterien/Flexible Definition
- Stärkere Abbildung von Substanzeigenschaften im Verfahren

Es wurde diskutiert, ob substanzspezifische Eigenschaften (z.B. Wirkmechanismus, e-fate Eigenschaften) ausreichend im RAC abgebildet werden? Zu diesem Punkt gab es – auch im anschließenden Plenum – kontroverse Meinungen. Die Berücksichtigung von Fate und chronische Effekten wird insbesondere für Mehrfachapplikationen als notwendig erachtet. Ebenso wird die flexible Definition von Hotspot-Kriterien gewünscht, um auch hier substanzspezifisches Verhalten berücksichtigen zu können.

Vor dem Hintergrund der Planungssicherheit wurde die Frage diskutiert, ob die Aktualisierung der Geodatenbasis einen neuen Stand von Wissenschaft und Technik darstellen kann?

- Einführung des Verfahrens stellt einen neuen Stand von Wissenschaft und Technik dar. Hier sind Übergangsfristen erforderlich um Planungssicherheit zu gewährleisten.
- Die Aktualisierung der Geodatenbasis stellt keinen neuen Stand von Wissenschaft und Technik dar. Dies wurde kontrovers diskutiert.
- Grundsätzlich besteht der Bedarf, während der Dossiererstellung rechtzeitig über eine Aktualisierung der DB informiert zu werden.

Zum Ende der Sitzung könnte die geäußerte Forderung nach Zugang zu harmonisierten Geodatenbestand und Werkzeugen für Berechungen an den Rechnern der Antragsteller nicht mehr näher beleuchtet werden.

AG 6: Übertragbarkeit auf Feldkulturen – Risikoabschätzung und Managementoptionen

Leitung: Martin Bach und Michael Klein

Teilnehmer/-innen: G. Berger, M. Kerber, J. Krumpe, S. Matezki, M. Morgenstern, A. Osterwald, D. Rautmann, C. Schriever, H. Tischner

Notwendigkeit für georeferenzierte PRA-Drift für Feldkulturen wird aus verschiedenen Gründen *wesentlich* geringer angesehen im Vergleich zur geoPRA-Drift für Raumkulturen:

- Zeitfenster für Behandlungen ist im Regelfall groß (Herbizid-Behandlungen bspw. mehrere Wochen); das ergibt sich unter anderem bereits aus der wesentlich geringeren technischen Behandlungskapazität der Landwirtschaft im Ackerbau (ausgedrückt in Feldspritzgeräten pro hundert Hektar Behandlungsfläche).
- Driftdepositionen (BBA-Eckwerte) sind bei Balkenspritzgeräten wesentlich geringer im Vergleich zu Raumsprühgeräten (mit Trägerluftstrom).
- Einheitlichere und engere Abstandsmatrix als bei Raumkulturen
- Technischer Aufwand und Kosten für driftmindernde Technik sind bei Balkenspritzgeräte in Form von Drift-reduzierenden Düsen wesentlich geringer und finden daher eher Verbreitung.
- Notwendigkeit einer Vereinheitlichung und Vereinfachung von Abstandsauflagen bzw. einer Reduzierung der Spritzabstände im Ackerbau wird derzeit von keiner der beteiligten Seiten für dringlich erachtet.
- offensichtlich nur wenige PSM mit einer aktuellen 20 m-Auflage belegt (siehe Übersicht von Herrn Morgenstern während der AG)

Andererseits ist die unterschiedliche Landschaftsstruktur in Regionen mit Ackerbau zu berücksichtigen: insbesondere in Nordost-Deutschland mit seinen z.T. sehr großen Ackerschlägen könnte die Situation auftreten, dass ein einziges Feldstück über mehrere hundert Meter an ein Gewässer angrenzt. Diese Gewässerstrecke würde dann innerhalb einer vergleichsweise kurzen

Zeitspanne (Fahrdauer der Feldspritze) zusammenhängend behandelt und könnte von Driftdeposition betroffen sein.

Bei Einführung geoPRA in Sonderkulturen könnte es passieren, dass ein Wirkstoff eine andere Abstandauflage für die Anwendung in Sonderkulturen bekommt als in Feldkulturen. Dies wird jedoch nicht als ernsthaftes Problem für die Akzeptanz der Auflage auf Seiten der Landwirtschaft gesehen und könnte zunächst auch erst einmal mit der unterschiedlichen Driftstärke von Flächen- und Raumspritzgeräten begründet werden.

ABER: Bei der PSM-Anwendung in Flächenkulturen stellen Drifteinträge nach Stand der Kenntnis nicht das zentrale Problem dar. Die mengenmäßig bedeutendsten Eintragspfade sind mutmaßlich Runoff und Dränage. Das Risiko dieser Eintragspfade wird derzeit mittels EXPOSIT bewertet, was jedoch keine georefenzierte, probabilistisch basierte Expositionsberechnung ermöglicht. Differenzierte Modellansätze zur Abschätzung von PSM-Runoff- und Drainage-Einträgen in Oberflächengewässer sowie zur Lokalisierung von Risikoflächen im Landschaftsmaßstab stehen im Prinzip zur Verfügung. Zeitpunkt und Intensität von PSM-Gewässereinträgen über den Transportpfad Runoff (und Erosion) hängen dabei von einer Vielzahl von Faktoren ab, die Prozessabbildung ist erheblich komplexer als bei Drift. Im Vergleich zu einer geoPRA-Drift würde eine geoPRA-Runoff daher wesentlich mehr probabilistische Elemente enthalten; aufgrund räumlich und zeitlich hochvariabler Eingabegrößen wären die Ergebnisse mit deutlich größeren Unschärfen behaftet.

Anlage 4: Protokolle aus den Plenardiskussionen

Plenardiskussionen 16. November 2009

Rapporteur C. Schäfers

Einführung Workshop: Jörn Wogram

Einführung Thema: Roland Kubiak

Grenzen wissenschaftlicher Betrachtung ausgelotet

Fließgewässer Expo – Ökotox

Mitarbeit gefordert

Steffen Matezki: Historie des Themas (Folien s. Anhang)

Seit 2002 Geodatenbasis-Aufbau

Seit 2006 Geo-PERA

Workshop 2007 Landau: Report als UBA-Text (47/08), Voraussetzung für Projekt

Ergebnisse von Landau dargestellt

Udo Hommen: Übersicht über GeoRisk-Ansatz (Folien s. Anhang)

Verständnisfragen:

IVA Dechet: 95 - vs 90 – Perzentil?

Hommen: 95 für Populationsschutz, 90 für Exposition / Belastung

Wogram: „an Sicherheit grenzende Wahrscheinlichkeit" **überall** schädliche (nicht vertretbare) Effekte ausschließen (95 %)

Hommen: Recovery macht Effekte vertretbar, Überall heißt: Lokale Population (Hotspot = RMA: 1000 m), aber nicht in jedem einzelnen Segment

Liess (?): Effekt-Probabilistik, SSD?

Hommen: RAC kann auf Basis SSD abgeleitet sein. Aber nicht unterschiedlich zu jetzigem Verfahren

Michael Klein: AP1 (Folien s. Anhang)

Verständnisfragen:

Berger: Warum Windrichtungen gleichverteilt?

Klein: Oberflächennah keine Vorzugswindrichtung

Probabilistisch: Gesamt-worst-case-Perzentil

Streloke: Richtung gegengleich worst worst case, Nur eine Himmelsrichtung links und rechts

Klein: Für Einzelberechnung eine Richtung, aber schlechteste Richtung per Perzentil berücksichtigt

Streloke b) Sedimentkonzentration: Ökotox. Womit verglichen?

Hommen: OECD 218 (Chironomus-test), wie im jetzigen Verfahren

Wogram: Trigger Daphnientox + WS-Verlagerung: => *Chironomus* wie im jetzigen Verfahren

Rautmann: Begriff „Driftreduzierende Technik“ statt „Düsen“ verwenden,
Windstärke-Effekte theoretisch? (ja)
Wind vom Gewässer weg?

Klein: wird probabilistisch gezogen und entfällt bei Perzentilbetrachtung

Wogram/Rautmann: Bei Windstille weiß man nichts

Streloke: Abdriftmessungen immer bei „richtiger“ Windrichtung, keine Info über reale Situation im Vergleich

Golla: Für jeden Richtungsstrahl berücksichtigen wir die Abdriftsituation der Versuche

Morgenstern: Vorherrschende Windrichtung sollte berücksichtigt werden (s. Herbert Koch-Experimente).

Golla: Windsituation in Obstanlagen schwer zu definieren. Gleichverteilung erst einmal default. Jederzeit veränderbar, wenn neue Erkenntnisse

Resseler: Abschirmung generisch; Emerse Vegetation nicht: Falsch

Klein: Mahd

Hommen: Generisch worst case, Hecke ohne Laub, da substanzunabhängig (konservativ)

Resseler Applikation nach Perzeptionsfaktor, ohne Laub ist meist unrealistisch

Burkhard Golla: AP2, Hotspot mit statischem Gewässermodell (Folien s. Anhang)

Verständnisfragen:

Schad: Für jedes Segment 90-Perzentil der Expo abgespeichert? Von Segment zu Segment immer Windrichtungswechsel zur ungünstigsten Situation. (Bestätigt).

CS: erhöht 90% Sicherheit signifikant

Burkhard Golla (a), Matthias Trapp (b): Datenunsicherheit (s. Anhang)

Verständnisfragen (a):

Streloke: Unterschiede zwischen den Ländern, warum? Wozu der Vergleich FB/ATKIS überhaupt?

Golla: Unterschiedliche Auftraggeber/-nehmer/Technik
Anspruch bei Gewässerschutz überall ist viel größer als bei Landschaftsperzentil. => Info über Unsicherheit bei der Erfassung expositionsrelevanter Flächen unbedingt nötig.

Verständnisfragen (b):

Kubiak: Genauigkeiten, Ungenauigkeiten in ATKIS: Wie ist die Meinung?

Matties: Breite, Tiefe: wie?

Trapp: Noch nicht beschrieben, bislang nur statisch 3.33-Verhältnis

Matties: PEC ist nicht georeferenziert, wenn Volumen nicht geo-referenziert!
PEC ist immer volumenbezogen. Nur Eintragsweg ist geo!! Deutlich machen.

Kubiak: statischer Ansatz ist 1. Schritt, echte Georeferenzierung erst später

Golla: Expositionsabschätzung soll verändert werden. Referenzdatensatz erst mit Standardgewässer

Matties: Nicht überall wo Ihr ATKIS draufschreibt, ist geo drin.

Schad: PECmax bleibt gleich ATKIS->HR, kaum Hotspot-Verringerung
Nach Schulz soll Faktor 10 sein. Wie soll verfeinert werden?

Kubiak: Unterschiede größer bei geringerer Instensität/größerer Diversität der Nutzung.

Trapp: Hallertau mit großen Unterschieden, sobald Anstaände größer/Nutzungen weniger intensiv.

Golla: (*nicht verstanden)*

Dechet: Wind: Zeitlicher Anteil der Applikationen unterhalb der Versuchsgeschwindigkeiten wichtig?

Golla: Fahrtgeschwindigkeit wird dann wichtig; keine Daten

Dechet/Streloke: Grundlage der Deposition bei Windgeschw zw. 3 und 5 m. Ist das realistisch?

Wogram: Depositionsrate von Windgeschw. Nicht nachgewiesen

Rautmann: aber nur zwischen 3 und 5 m, dehalb nicht nachweisbar. Dazu heftige Schwankungen in Richtung und Geschwindigkeit

Berger: Rautmann-Tabelle Sinn?

Rautmann: Worst case für Deterministik. Konservativ

Morgenstern: Konservativität muss erwähnt werden.

Berger: Heckenabschirmung: 25% 75% wann? Jahreszeitliche Auflösung (Bestätigt)

Hommen: Generisch 75% kulturspezifisch möglich (wenn frühere Applikationszeiten ausgeschlossen)

Schriever EVA Eintragspfad Verflüchtigung generisch?

Klein: Nein substanzeigenschaftspezifisch.

Martin Bach: Fließgewässeransatz (s. Anhang)

Dispersion relativ unwichtig, Entscheidend ist Behandlungswahrscheinlichkeit, Behandlungszeitfenster, PECtwa-Pezentil

Verständnisfragen:

Rautmann: Zeitlicher Verlauf der Applikation: Auf einmal, mit/gegen Fließrichtung

Bach: Bei PECtwa über eine Stunde ohne Effekt.

Matezki: Korrelationen zwischen Segmenten (mehrer Segmente zu einem Feld)

Bach: Zu untersuchen. Im Obstbau bei Anlagengrößen wohl kein Problem (zu klein); im Feldbau anders

Trapp: Fließgewässermodell aus ATKIS (s. Anhang)

Schad: Applikationsfenster wie lang? Applikation wann?

Trapp: Dauer: Vorbeiflussdauer; Zeit: Alle gleichzeitig

Kubiak: Diskussion Fließgewässer

Ressler: Zeitfensterbetrachtung 6 µg/L mit Maximalabstandsauflage

Dechet: Konzentration im kleinen Zeitfenster hoch = worst case. Was ist mit längerer Exposition bei niedriger Konz.?

Kubiak: Nähert sich altem Szenario

Hommen: Time over threshold addieren

Streloke / Wogram: bei längerem Expozeitfenster möglicherweise höheres Risiko.

Kubiak: Vertagung auf morgen

Matties: Breite / Tiefe am wichtigsten (bestätigt); ist variabel (abflussabhängig).

Trapp: Gewässertiefe (bestimmt Verdünnung)

Morgenstern: Fließgeschwindigkeit (Verdünnung) oder Breite (Eintragsmenge)?

Schad: Echte Probabilistik in diesen Überlegungen drin, sehr posiiv

Streloke: Behandlungsintervall: Erregerabhängig. Bei Pilzen alle sehr gleich (2d), sonst unterschiedlich.
ATKIS-Breite?

Antwort: Wird nicht mehr genutzt, wie umgehen mit Schwankungen?

Schad: Verteilungsfunktionen statt worst-case zum Anwendungsmuster (zB aus empirischen Erhebungen)

Bach: Hydraulik median betrachten (kein Perzentil)

Plenardiskussionen 17. November 2009

Rapporteur M. Roß-Nickoll

Zusammenfassung der Kernaussagen des 1. Tages (R. Kubiak, s. Folien im Anhang)

95-Perzentile zur Schutzzielerreichung: Keine unvertretbare Effekte auf lokale Populationen

90-Perzentile aus Expositionsverteilung (PEC) trägt dazu bei

- Unsicherheit bei Geodaten (z. B. Flächennutzung, Abstände Gewässer – Kultur)ist zu minimieren

- Systematische Betrachtung aller Verfahrensunsicherheiten

Statischer Ansatz ist nicht georeferenziert, da Volumen nur nach Standardmodell berechnet wird

Die PEC ist immer volumenbezogen: => Nur Einträge werden georeferenziert (und probabilistisch) betrachtet

Fließgewässer: Breite bestimmt Eintrag; Tiefe bestimmt Volumen (Verdünnung, PEC)

Dispersion relativ unwichtig

Entscheidend: Behandlungswahrscheinlichkeit, Behandlungszeitfenster, PECtwa-Pezentil

PEC-seitig: Zeitfenster entscheidet über „Schutzniveau“

=> Information über LW-Verhalten versus Vollprobabilistik

=> Zeitfenster Pestabhängig (Fungizide)

Wirkliche Verteilung der Windrichtungen spielt bei Nutzung der 90-Perzentile keine Rolle

Die generische Hotspot-Ermittlung kann kulturspezifisch erfolgen, da RAC, Hecken, ufernahe Krautvegetation kulturspezifisch betrachtet werden können: Obst früh, spät, Weinbau, Hopfen

Refinement ist substanzspezifisch

HR/ATKIS-Vergleich:

Unterschied steigt mit geringerer Nutzungsintensität (und ganz nah am Gewässer – Interpretation HR, Genauigkeit Atkis)

Wogram: Vergleich Unsicherheiten – Verteilungen;

Matties: Frage besser mit best estImate beginnen und am Ende erst worst case;

Schäfers: deshalb mit Landschaftsparametern best estimate und bei anthropogenen Parametern erst worst case;

Michael Klein: Im Moment bei Focus genau andersherum

Dechet: Vorschlag Schäfers;

Matties: bei Spraydrift eher worst case üblich

Wogram: Breite der Verteilung ist ausschlaggebend, entweder konservative Einzelschätzung oder Verteilung selbst;

Hommen: nur Windrichtung und Driftbelag probabilistisch, d.h. ähnlich Vorschlag Wogram

Liess: Schutzziel lokale Populationen, welche adaptierte, welche Gemeinschaft ist die Basisgemeinschaft?

Wogram: Referenz nicht Ist Zustand sondern gesetzlich vorgeschriebener Erhaltungszustand = realistic best case

Dechet: Verbesserungsgebot gibt für das Pflanzenschutzgesetz nichts her

Streloke: Grundaussage Dispersion ist vergleichsweise unwichtig ist allgemeingültig gilt immer, wird von Bach bestätigt

Matezki: Behandlungszeit könnte Szenarien basiert sein, most likely, best und worst case

Wogram: Wie konservativ ist Dispersionsschätzung Modell – Messung?; Streloke: chemisches Monitoring, zur Peakkonzentrationsmessung braucht man Ereignis gesteuertes Design; Ressler: Zuflüsse berücksichtigen (Kirche in der Röhre lassen?)

Ableitung von Hotspot-Kriterien (U. Hommen)

UBA Kriterium 2007: Wenn PEC>RAC, dann 100% Effekt

Carabaryl Acetylcholinesterase Hemmung: Steigung der SSD bei allen Organismen gleich => Dosis Wirkungsrelation => Abhängigkeit LC 50 von Belastungszeit (analog zu FOCUS) bei dynamischen PEC gemittelte Werte über eine Stunde Grundlage der Bewertung als PEC TWA/1h, angewendet auf Meilenhofen Bach keine kritischen Segmente??

- Generelle Vorgehensweise AP3
  - Identifikation von Traitgruppen bzw. Stellvertreterarten
  - Ableitung gruppenspezifischer Hotspot-Kriterien
- Bewertung von in den Fließgewässern zu erwartenden Kurzzeitexpositionen
- Konsequenzen für die Ableitung von RMS

Berger: Dosis Zeit hohe Peaks kurze Zeit => Fläche unter Kurve als Maß nehmen, geht nicht wegen nicht nur kummulative Beziehung sondern konservativer; Wogram: Mittelwert Integral über Zeit => Spitzen werden wegnivelliert (Wirkungsreziprovität, Udo: stimmt nicht da logarithmische Skala)

Liess: Literatur einbeziehen, Lindan...; => Substanzabhängigkeit!

Wogram: nicht für jedes Gewässer ist ein h TWA typisch; Konzept ERC sollte angewendet werden, für welches Gewässer ist welches Expositionsmuster typisch,

Schäfers: Summierte Expositionszeiträume => ERC 48h alles was drüber liegt geht in chronische Toxizität

Wogram TWA über 3h kann worst case sein in Vergleich zu andern Überschreitungen 8nicht verstanden)

Streloke: zu kompliziert, Gruppe Fungizide Dauer Effekt sehr heterogen=> keine stärkere Auftrennung der Toxbewertung im akut – chronischen Bereich

Ratte: Wie sollen higher tier Daten integriert werden??;

Hommen: Mesokosmos hat das Problem der Belastungsdauer nicht, dann RAC aus Spitzenkonz. Mit max TWA einsetzen => Methode zur Effektauswertung der FOCUS Szenarien?

Vorschlag zur technischen Umsetzung des geplanten Zulassungsverfahrens (B. Golla, J..Krumpe)

Rollenspezifische Handhabung UBA Sonderrolle, Wie könnten Mehrfachapplikationen integriert werden?

HR Daten werden nur erhoben um Hotspots zu reduzieren => zyklische Auffrischung der Datenbasis über Integration von HR Daten

- Ablauf einer Pflanzenschutzmittelprüfung (Antragsteller und Behörde)
- Programmoberfläche
- Aktualisierung des Datenbestandes / Historisierung
- Verfügbarkeit der Daten

Allgemeine Diskussion

Schad: Substanzspezifische Verfeinerung der Effektbewertung wird wie integriert?, Krumpe: Tool ist für statisches Modell entwickelt, Golla sieht Möglichkeit zur Erweiterung aber substanzspezifische Bewertung ist nicht ursprünglich Aufgabe dieses Tools

Spickermann: Stakeholder finanzieren Verbesserung dieses Datensatzes, das käme allen Nutzern zugute;

Schriever: Integration Zusatzdaten aber keine Neuberechnung der PEC Werte? Krumpe: Wäre nur nötig bei Veränderungen die die generische Analyse betreffen.

Matezki: Tool beinhaltet Fliessgewässerdynamik nicht wegen Historie des Projektes, aber modularer Aufbau nach Leisstungsbeschreibung;

Krumpe Tool ist immer nur der Konsens der AG, daher Frage nach Fliessgewässerdynamik hier

Golla: A Auftrag des modularen Softwareentwicklung ist angelegt, bei PECini statt PECTWA wäre das andere Integration für Information am Gewässerpunk

Schad: Substanzspezifische Hotspot-Bewertung ist neu? Hotspot-Analyse wird immer sehr dynamisch bleiben und neu gerechnet werden können müssen

Dechet: vorherige Übereinkunft, dass es keine substanzspezifische Bewertung geben darf

Kubiak richtig aber verteilt in 2 tier generisch und substanzspezifisch ist so wie bisher; kein Konsens, Schad: wie kann Refinement schritt integriert werden?

Praktikabilität, wer finanziert, was ist generisch zu machen, wie kann Refinement integriert werden?

Plenum mit Ergebnissen der AGs

Klein: mit Modell Runoff für jedes Feld generisch Michael möglich

Trapp: für jede landwirtschaftliche Fläche wird Erosionswert berechnet (bundesweites Kataster).

Bach: Verbindung zur WRRL : Grenzwerte für PSM, falls diese im Monitoring festgestellt werden, müssen die Wasseraufsichtsbehörden aktiv werden und die Eintragspfade ermitteln. Dazu Modellentwicklung für Eintrag aus drei Pfaden, Drainage, Runoff, Drift möglich => auch Faktor „Fehlverhalten“ sollte integriert werden.

Schad: im Ansatz wird die Verfeinerung mehr in der Komplettierung der Geodaten gesehen, im dynamischen Modell definieren die substanzspezifischen Eigenschaften wesentlich den Hotspot auf Seite der Effektbewertung. Solche Module bei Detektierung von Hotspots sollten im Tool zur Verfügung stehen.

Schäfers: Chronische Effekte bei niedrigen Konzentrationen müssen ins Verfahren integriert werden können, ist unzureichend betrachtet.

Plenardiskussionen 18. November 2009

Rapporteur U. Hommen

Zusammenfassung der Kernaussagen des 2. Tages (s. Folien Roland Kubiak)

- Bisher Erreichtes
  - Bundesweite Berechnung der PECs auf der Basis von ATKIS und „Standardgrabenmodell“: Georeferenzierte probabilistische Berechnung der Einträge, zu erwartende km MS je Kultur in D
  - Unterschiede ATKIS – HR größer in heterogenen, kleinräuniger strukturierten Landschaften
  - Grundlage gelegt für Erzeugung eines topologisch korrekten, gerichtetem Gewässernetz gelegt als Vorraussetzung für dynamische Fließgewässermodellierung
  - Sensitivitätsanalysen der Expositionsmodelle zur Indentifizierung der treibenden Parameter für die ´PECs
  - Konzentrationsverläufe für reale Beispielgewässer unter konservativem Annahmen berechnet
  - Entwicklung ökotoxikologischer Kriterien in Bezug auf Traitgruppen (im Aufbau)
  - Konzept für Wirkabschätzung kurzfristiger Exposition
  - Anwendungstool konzipiert und zum großen Teil fertig
- Einführung eines neuen Verfahrens unter Berücksichtigung der Fließgewässeransatzes sowie einer angepassten Bewertung der Kurzzeitexposition grundsätzlich möglich und von Workshopteilnehmern präferiert
- Anzahl Managementsegmente (MS) nach diesem Verfahren nach jetziger Einschätzung unter denen des statischen Standardgrabenmodells.
- Die nächsten Schritte

  - o Abschluss der jetzigen Projektes
  - o Festlegung der Parameter zur MS Verrechnung mit allen Beteiligten
  - o Berechnung der MS für mögliches Pilotprojekt Hallertau
  - o Entscheidung über Etablierung eines Pilotprojekts
  - o Durchführung des Pilotprojekts mit begleitendem chemischen und biologischem Monitoring
  - o Entscheidung über die Einführung des Verfahrens zunächst für alle Raumkulturen
- Projekt liefert Erkenntnisfortschritt, der laut PflSchG auch berücksichtigt werden sollte

Diskussion:

HR: Beeindruckende Ergebnisse, Module wurden kurz erklärt, für genaue Beurteilung muss detaillierte Dokumentation vorliegen. Ergebnisse einer RS Abschätzung müssten zunächst der Landwirtschaft kommuniziert werden, um Akzeptierbarkeit zu prüfen
MS: Ergebnisse müssen auch in die wissenschaftliche Fachwelt (international) kommuniziert werden

RK: Geplant für SETAC Europe 2010

GB: Validierung (Überprüfung der RMS Ergebnisse in der Landschaft) notwendig, Bedeutung des Monitorings im Pilotprojekt

Klein: Win/Win Situation mit Naturschutz bedenken: je weniger RS, desto weniger Trade-off für Naturschutz

TS: Komplexes Gesamtvorhaben. Für die wesentlichen neuen Punkte exisitiert noch keine Erfahrung. Erheblicher Entwicklungsbedarf. Nach vorliegenden Ergebnissen zu früh für Entscheidung über Einführung des Verfahrens, dazu werden mehr Informationen benötigt

Schutzniveau für Oberflächengewässer (s. Einstiegsfolie von Jörn Wogram)

Gesetzliche Anforderungen: Dir 91/414 besagt: keine nachhaltigen Schäden an Pop von Nichtzielorganismen, Paraquat-Urteil: „mit an Sicherheit grenzender Wahrscheinlichkeit“

Schutzgut sind Populationen, nicht Individuen

Surrogatefür mit an Sicherheit grenzende Wahrscheinlichkeit: GeoRisk: 95 % Sicherheit für ein Jahr, Zulassungsdauer 10 Jahre, Spritzserien

Surrogat für Population: 1000 m Gewässerabschnitt, Traitkombi, 10 % Mortalität pro Abschnitt und Jahr tolerabel

Anforderung Schutzniveau hoch zu halten bleibt bestehen, theoretisch könnte GeoRisk zeigen, dass das alte System nicht genügend protektiv war, das Maß ist nicht uneingeschränkt das alte Bewertungssystem

Diskussion:

CS: GeoRisk bietet Möglichkeit der Frachtberechnung, wichtig für andere Regelwerke, z.B. Wasserrahmenrichtlinie

SM: Hotspot-Management kann in Beziehung zu anderen Regelwerken stehen, kann auch Ziele anderer Bereiche bedienen

TS: „10 % Effekt“ statt „10 % Mortalität“?

JW: Mortalität immer konservativ, generisches, nicht trait-basiertes Kriterium!

UH: Ziel auch in Hotspots sind max. 10 m Abstandsauflage das Ziel, Option dann in Einzelfällen Rücknahme der Kultur auf > 10 m, wenn keine anderen Maßnahmen möglich sind

FD: Präferenz des IVA für die Möglichkeit von Abstandsauflage bis 20 m in Hotspots

MS: Risiko, dass Wegnahme der Kultur für viele Mittel überprotektiv ist und zu Verlust von Kulturfläche führt

GB: Wegnahme von Kulturfläche kann nicht die Option sein, sondern Reduktion des Eintrags, fatal für Bauer die Auflage zu fordern, dass er 20m seiner Kultur zürücksetzt, es gibt andere Maßnahmen

MaK: Generisches Modell weist x km MS aus, diese würden gemanaged

MS: Managament kann doch nicht auf 1 – 2 Worst Case Mittel aufbauen

JW: Doch, das ist der Grundansatz: Aussage der Landwirtschaft, wir brauchen die worst case Mittel

HR: 4 -5 Firmen mit jeweils 3 bis 7 Mitteln mit 20 m Auflage, Ein nicht Einhalten von Auflagen kann ordnungspolitisch kein Grund sein, die 20 m abzuschaffen

AM: Noch kein Konsens zur Interpretation von 20 m Abstand in Hotspots

UH: Verbreiterung der Uferstreifens muss ja nicht von 10 m direkt auf 20 m sein.

MT: Alle Optionen ausschöpfen, um ohne Verbreiterung auf 10 m zu kommen (Schilfgras?)

TS: Wann kommt man denn auf akzeptierte lokale Effekte: Effizienz des Toleranzkriterium? Die Toleranz von Überschreitungen ist eine der Grundlagen des Verfahrens und wird betont. In der Kulturlandschaft wird vermutlich ein einzelnes Feld in Gewässernähe ausreichen, einen Hotspot anzuzeigen, bzw. umgekehrt jedes Gewässer, das durch Kulturen fließt wird vermutlich zu Hotspot führen. Diese werden dann Risikomanagement unterzogen. Wo ist effektiv die Toleranz von lokalen Überschreitungen?

MG: Wirkt sich das Verfahren nur für 20 auf 10 aus oder auch von 10 auf x aus?

SM: Ja, prinzipiell sollte es auch zu geringeren Auflagen für jetzige 10 m Auflagen kommen

Unsicherheiten des neuen Verfahrens (Einstiegsfolien von B. Golla & M. Trapp)

- Unsicherheiten in der Abbildung der Applikationsflächen und Gewässer
  - ATKIS bildet Gewässer nicht vollständig ab – Einschätzung dieser Unsicherheit bis zum Ende des Vorhabens
  - ATKIS dennoch derzeit bestmögliche Abb. der Gewässer
  - InVeKoS auf Basis HR bilden Appl.flächen bundesweit ab.
  - Lösung der Ungenauigkeiten bei der Erfassung von Applikationsflächen durch Nutzung amtl. geprüfter landwirtschaftl. Fachdaten
- Unsicherheit der hydrologischen Eingangsparameter
  - B/T Verhältnisse
  - Fließgeschw / Abflussmenge (prinzipiell verfügbar)
- Zeitfenster der Applikation

Diskussion:

GB: Unsicherheiten bestehen nicht nur bei der Expositionsabschätzung,. Wie sind die Unsicherheiten bei der Ökotox?

UH: Unsicherheiten / Sicherheiten bei der RAC Ableitung aus dem jetzigen Verfahren werden beibehalten (Sicherheitsfaktoren bzw. Triggerwerte)

CS: Fließgewässermodell verschiebt Fokus auf akute Effekte, langfristige Effekte müssen über längere Strecken berücksichtigt werden

MKe: Invekos-Daten in Ländern verfügbar, können für Pilotflächen abgefragt werden

Für Applikationszeitfenster können NEPTUN Daten genutzt werden

MT: INVEKOS umfasst teilweise auch andere Landschaftselemente, allerdings nur in Zusammenhang mit landwirtschaftl. Flächen
Hecken können eventuell durch EU Mittel gefördert werden

MaK: Förderung landesspezifisch, Förderung gleicht Produktionsverlust nicht aus

GB: Förderung nur von sommergrünen Laubhecken möglich

Managementmaßnahmen an potenziellen Überschreitungssegmenten (Einstiegsfolien Martin Bach)

- Applikationstechnik
- Behandlungsmanagement
- Landschaftsbezogene Maßnahmen

  Rodung gewässernaher Reihen

  Anpflanzen von Hecken

  Anlage von naturnahen Uferstreifen

  Verschieben der Böschungsmahd

  Unterlassen der Böschungsmahd

  Technische Maßnahmen: Driftschutzzäune

  Higher Tier: Verbesserung der Wiederbesiedlung / Anlage von Regenaritionsräumen

  - Aspekte

  Wirksamkeit

  Träger, Veranlassung

  Synergien

  Kosten (Höhe, Träger)

  Organisation / Durchführung

  Kontrolle / Gewährleistung

Vergleich von PSM mit den Auswirkungen von Pflegemaßnahmen an Gewässern

Diskussion:

GB: Wie effektiv sind Driftzäune? Wieso nur als Übergangslösung?

MKe: Ästhetischer Aspekt ist zu bedenken

GB: Mehr Flexibilität der Managamentoptionen!

JW: Ja, auch in Bezug auf Hecken, z. B. Driftminderung in Abhängigkeit von der Breite der Hecke

MB: Kaum Wissen über Effekte der Gewässerunterhaltung, eventuell Traditionsverhalten der Verbände / Erfüllung des Auftrags. D.h. oft ist die Pflege so nicht notwendig, verminderte Abflussleistung oft kein Problem (eher besser für Hochwasserschutz). Weniger Pflege ökologisch besser: Gewässerunterhaltungsmaßnahmen integrieren!

MT: Ökologischer Effekt von Renaturierung ?

RK: Finanzierung / Durchsetzbarkeit von Maßnahmen?

MS Wenn Management wg Zulassung, dann keine EU - Förderung. Wenn außerhalb der Zulassung, dann kein Druckmittel

SM Hängt ab von der Gestaltung der Übergangsphase

JW Parallelen zwischen Hotspot-Management und Flurbereinigungsverfahren: Beides Eingriffe in die Landschaft mit Ziel der Produktionsoptimierung?

GB Flächentausch bei staatl. übergeordneten Verfahren machbar und gerechteste Option, aber sehr langfristig

RMK Flurbereinigung auf freiwilliger Verfahren möglich?

MT nur da möglich wo noch keine Flächenneuordnungsverfahren durchgeführt worden sind

MB: Nicht Flurbereinigungsverfahren durchführen, sondern aus der Art des Verfahrens lernen (FBV als juristischem Präzedenzfall)

BG: Im Auge behalten, dass auch Synergien mit anderen Einträgen / Stressoren entstehen können

CS: Übertragbarkeit von Verfahren zur Ausweisung von Wasserschutzgebieten

MT Bei breiteren Hecken höhere Driftminderung anwendbar?

JW/SM Im Prinzip ja, wenn belegbar

Meinungsbild des Plenums zum neuen Ansatz

IVA-GeoPERA Positionn zu GeoRisk (s. Folien von F. Dechet)

- Grundsätzl. Position zu GeoPra: Unterstützung von GeoPra, generischer Hotspot-Identifizierung, und Risikomanagement in der Landschaft
- Landschaftsbezogenes Risikomanagement und Zulassung sollen aber getrennt sein: konzeptionell und verfahrenstechnisch
- Landschaftsbezogenes Management kann nicht Auflage im Zulassungsverfahren sein
- Zulassung sollte auf Basis einer bundesweiten Verteilung durchgefügt werden
- Hotspot-Analyse sollte nicht Grundlage der Zulassungsentscheidung sein
- Über „geoEckwert Aquatik“ könnten die Resultate von PRA als Weiterentwicklung der bisherige Drifteckwerte eingefügt werden
- Die Analyse muss beim Antragsteller laufen können
- Vorgehen
    - Engere Einbindung von Landwirtschaft und Industrie

- Koordinierung aller Aktivitäten durch eine verantwortliche Stelle, z.B. durch BVL-Beirat
- Praktikabilität muss in Pilotstudien überprüft werden
- Erstellung eines Ablaufplans für Implementierung

Diskussion:

JW: Wenn Zulassung kein neues Hotspot-Management notwendig macht, was ist der Unterschied zu GeoRisk?

SM: Ist der Aufwand nicht vergleichbar?

TS: Generische Hotspot Analyse wird nicht angetastet, nur Zulassung soll davon unabhängig sein. Der in GeoRisk vorgeschlagene Ansatz ist sehr komplex. Die Zulassung hängt an Vorhersage eines Hotspots.

UH Auch bei GeoRisk soll generisch von Zulassung getrennt sein.

FD Generisches Risikomanagment könnte durch Nationalen Aktionsplan (NAP) erreicht werden.

JW Zustimmung, dass mit NAP koordiniert werden soll (Synergie), Ziel NAP ist allerdings Risikosenkung nicht Herabsetzung der Auflagen bei Sicherstellung des Schutzniveaus

UH IVA Vorschlag macht Traitspez. Kriterien überflüssig,

MaK: Ausgestaltung der Landschaft ist nicht Aufgabe der Dienste, deren Aufgabe ist Beratung und Kontrolle. Der neue Ansatz bietet keine große Veränderung für die Aufgabe der Überwachung der Landwirte
Vereinfachungen des Verfahrens wäre jedoch eine Erleichterung der Beratung

MB Wie könnte der BVL-Beirat „re-aktiviert“ werden?

FD Mehrere Akteure, keiner mit „Vogelperspektive“

RK Es gibt auch noch das Lenkungsgremium…

MB Wenn Zulassung nur auf Geo-Abdrifteckwerte, wozu dann wird dann vollständige Information gebraucht

TS Weil die Geo-Abdrifteckwerte daraus abgeleitet werden

MS Wenn ein Mittel nicht die GeoWerte erfülllt, soll es dann Higher Tier Expositionsverfahren geben?

TS Es sind Risikominderungsgruppen vorgeschlagen (in BVL-Beirat diskutiert)

JW UBA größtmögliches Interesse an Transparenz und Fortschritt. Alleingänge nicht auf der Ebene der Fachbehörde, sondern auf politischer Ebene

FD Insgesamt starker Wunsch nach aktiver koordinierender Stelle

Abschluss:

RK Dank an UBA für offene Zusammenarbeit und Diskussion, Dank an Beirat für konstruktive Diskussion und Hilfestellungen
Dank an A Müller für Organisation des Workshops vor Ort
Dank an Workshopteilnehmer für Verdauung der harten Kost der ersten Tage und die sehr aktive Mitarbeit in den Breakout-Sessions und den Plenardiskussionen

JW Fazit aus Sicht des UBA

Konsens über

- Dynamische Expositionsmodellierung
- Trait basierte Effektbewertung
- Georefernzierte Risikoabschätuzung
- Roadmap zur weiteren Entwicklung

Outlook

- Drawback politischer Handlungsbedarf
- Unsicherheit über den Aufwand des Hotspot-Managements
- Transparenz de

Dank an die Teilnehmer für Sachlichkeit, Fachkunde, Verantwortlichkeit

Dank an das Konsortium für professionelle Vorbereitung, Durchführung und gute Vermittlung des Projekts

Ende des Workshops

Anlage 5: Inhaltsverzeichnis des Anhangs mit den Präsentationen

Die Präsentationen sind in einer separaten Datei zusammengestellt.

1. Die Ergebnisse des UBA Workshops von 2007 (S. Matezki)
2. Ein neues Konzept zur Beurteilung von Drifteinträgen aus Raumkulturen in Oberflächengewässer (U. Hommen)
3. Berechnung der Drifteinträge und PEC-Berechnung Standardgewässer (M. Klein)
4. Ergebnisse der Abschätzung von Hotspots und Managementsegmenten auf der Basis des Standardgewässermodells (B. Golla)
5. Unsicherheiten der Geodaten (J. Krumpe, M. Trapp)
6. Die Einbeziehung von Fließgewässern in das Verfahren – Grundlagen und Datenanforderungen (M. Bach)
7. Die Einbeziehung von Fließgewässern in das Verfahren – Erste Umsetzung (M. Trapp)
8. Zusammenfassung des 1. Tages (R. Kubiak, C. Schäfers)
9. Ableitung von Hotspot-Kriterien (U. Hommen)
10. Vorschlag zur technischen Umsetzung des geplanten Zulassungsverfahrens (Burkhard Golla, Jens Krumpe)
11. Folien Zusammenfasung Tag 2 (R. Kubiak)
12. Schutzniveau für Oberflächengewässer (J. Wogram)
13. Unsicherheiten des neuen Verfahrens (B- Golla)
14. Managementmaßnahmen an potenziellen Überschreitungssegmenten (M. Bach)
15. Stellungnahme des IVA zum neuen Ansatz (F. Dechet)

GeoRisk

Geodata based Probabilistic Risk Assessment of Plant Protection Products (Georeferenzierte Probabilistische Risikobewertung von Pflanzenschutzmitteln)

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Appendix D

Recovery of freshwater populations after stress - a review of case studies in the context of pesticide risk assessment

Authors

Andre Gergs, Silke Classen, Udo Hommen, Tido Strauss, Hans-Toni Ratte, Thomas G. Preuss

November 30, 2010

Abstract

Environmental risk assessment for plant protection products is developing from worst case to more realistic approaches. These approaches involve extrapolation of effects between organisms and various exposure scenarios as well as exposure based on FOCUS-scenarios or even geo-referenced data. To achieve a more realistic way of risk assessment, understanding of the recovery potential of populations under field conditions is essential.

Therefore a literature study was carried out to assess the recovery potential of aquatic algae, invertebrates and fish following disturbances. Various types of stressors were considered, i.e. pesticides and metals as well as physical disturbance, flood, drought and restoration of stream and ponds. Selection of case studies was based on available descriptions of effects and recovery as well as the type of stressor. Case studies were grouped according to type of disturbance, class of water body as well as taxonomic group and the endpoint investigated.

Lentic zooplankton and phytoplankton recovered in nearly all cases within one year even after pronounced effects. Fast recovery was observed for algae followed by Ephemeroptera and Diptera independent of ecosystem type. Lowest recovery potentials were found for Trichoptera, Odonata, Plecoptera and Crustacean in lotic systems. In lentic systems benthic crustaceans, Trichoptera, Mollusca and Coleoptera did mostly not recover within one year. In general recovery of lotic macroinvertebrates appeared to proceed faster than recovery in lentic systems likely due to drift of organisms from undisturbed upstream reaches.

In most cases, it could not be figured out whether recovery was autogenic or allogenic. However, for several taxa with only one reproduction period per year (e.g. fish) the cases of fast recovery indicate that active movement was the main pathway of recovery while for taxa with high population growth rates (algae, zooplankton, some multivoltine macroinvertebrates) recovery can likely be explained by population growth alone.

The data included in this review clearly indicates that lotic and lentic systems differ in their potential for recovery and thus have to be differentiated within risk assessment. For prediction of recovery within the risk assessment of pesticides three main factor groups have to be taken into account: (1) stressor related factors (2) species related factors (3) habitat and landscape structure related factors. Whereas stressor related factors (e.g. dissipation times and the intrinsic toxicity to some standard test species) form the basis of current risk assessment, species related factors like life cycle properties are only basically considered. Trait based approaches might help in the future to cope with the problem of extrapolating effects and recovery patterns between species. For ecosystem related factors, the spatial heterogeneity of landscapes (e.g. presence, position and quality of refugia, presence of sources for recolonisation) has to be taken into account for an adequate prediction of recovery.

Keywords: recovery, recolonisation, plant protection products, freshwater communities

Introduction

Environmental risk assessment for plant protection products is developing from worst case to more realistic approaches, e.g. by the use of exposure scenarios covering different representative combinations of climate and soil conditions (FOCUS 2001) or by spatial explicit exposure modelling. Modelling exposure more realisticly asks to answer the question about what effects the resulting exposure pattern will have on the population level in aquatic organisms. The presented review is part of a project a probabilistic georeferenced approach for the registration of pesticides in Germany. Based on former projects and a workshop held at the Federal Environmental Agency (UBA) in Dessau (Schulz et al. 2007, 2009), the approach is first developed and should be tested for spray drift in permanent crops, such as vineyards, fruit orchards, and hops. Until now, the aquatic risk assessment for the regulation of plant protection products in Germany is very similar to the assessment done for the active substances at the EU level. One important characteristic of both frameworks is that the first tier risk assessment is conducted for a standard lentic water body of 1 m width and 30 cm depth with the wind blowing from the crop to the water body and no drift-mitigating vegetation between the crop and the water body. Using this standard scenario, Predicted Environmental Concentrations (PECs) due to spray drift inputs are calculated for different distances between the sprayed crop area and the water body to determine mandatory minimum spraying distances necessary for registration of the product in Germany, e.g. 3, 10 or 20 m. Alternatively or in addition, the use of drift reducing application techniques can be necessary to reduce the risk to acceptable levels.

However, in reality the wind is not always blowing directly from the crop to the water body, the water bodies are variable in width, depth and flow velocity, and there might be drift mitigating vegetation between crop and water body. Thus, in reality there is a large amount of spatial and temporal variability in the exposure situations, which cannot be reflected with a single standard scenario. Therefore, the German Federal Environment Agency (UBA) has decided to explicitly consider the variability in the landscape in order to establish a more realistic risk assessment. By being more realistic, the approach should result in less complicated label instructions. On the other side, the risk assessment must still be sufficiently protective for the environment.

The new approach is based on a geo-referenced probabilistic exposure calculation for all relevant water body segments (10 – 25 m) in Germany. For each of these segments the drift load and the initial PECs are calculated in a partly georeferenced (e.g. distance to crop), partly probabilistic way (e.g. wind direction at time of application). The percentile of the PEC distribution for each segment is then compared with the Regulatory Acceptable Concentration (RAC) derived from ecotoxicological tests. If the PEC exceeds the RAC, the segement is considered as a potential Risk Managament Segment (RMS) where local risk mitigation measure may be implemented to reduce the drift entry and thus, the risk. However, exceedance of the RAC in water body segment must not necessarily result in an unacceptable effects because the amount of the RAC exceedance (and thus, the magnitude of effect) and the spatial aggregation of RAC exceedance determine the effect and the potential of recovery of the local population. Thus, one critical step in the project is to develop criteria identifying aggregations of RMS which pose unacceptable risks to the local population. For example, a preliminary and pragmatic criterion to identify those 'hot spots' could be (UBA 2007): exceedance of the RAC in more than 10 % of the water body segments in a 1000 m section or exceedance of the RAC by more than a factor of 10 in at least one segment (that for very high PECs effects are not restricted to the single segment). It is obvious that for different species such a hot spot critetion could be different, depending for example on the intrinsic growth rate, the mobility and the dispersal of the species. Because it is not possible to define criteria for all the species which might occur in edge-of-field-water bodies in the agricultural landscape, it was decided that a trait based approach should be used.

A trait in this context is a specific property of a species, i. e. life cycle characteristics, feeding type, mobility, etc. The basic idea behind the trait concept is that different ecosystems (e.g. small streams in different landscapes, i.e. North and South Europe) might have different species. However, the whole species community of a given ecosystem type usually includes typical and comparable combination of traits. Thus, based on traits instead of species extrapolation of effects from one (model) ecosystem to another might be possible. For more general information on the use of the trait concept in ecotoxicology and environmental risk assessment see for example Usseglio-Polatera et al. (2000), Poff et al. (2006), Baird & van den Brink (2007), Baird et al. (2008).

Recovery has become an important concept for the evaluation of model ecosystem studies in the pesticide registration framework. According the EU Guidance Document on Aquatic Ecotoxicology “the NOEAEC (No Observed Ecologically Adverse Effect Concentration) is defined as being the concentration at or below which no long-lasting adverse effects were observed in a particular higher-tier study (e.g. mesocosm). No long-lasting effects are defined as those effects on individuals that have no or only transient effects on populations and communities and are considered of minor ecological relevance (e.g., effects that are not shown to have long-term effects on population growth, taking into account the life-history characteristics of the organisms concerned). Different recovery rates may therefore be acceptable for different types of organisms (SANCO 2002). However, the extrapolation of recovery from these kind of experiments to the situation in the field is still a matter of debate.

The aim of this review is to summarize literature data on field and model ecosystem studies where recovery of aquatic organisms after pesticide stress was monitored. The following questions should be figured out:

- Which taxa can recover in which time frames after a disturbance?
- What are the mechanisms driving recovery for the different taxa?
- Are the data provided in literature suitable enough to consider recovery within a GIS-based risk assessment?

Therefore the open literature was reviewed with a focus on case studies of recovery after the application of pesticides. To increase the database also other stressors were included, namely other chemical stressors (organic and metals), physical disturbance, drought and flood as well as the new development of ponds and streams. The case studies will be presented and discussed as far as the different disturbances are comparable to pesticide exposure. Additionally, a pattern of recovery was established for the different ecosystems and factors of disturbance.

Disturbance, stressors and recovery – some definitions

Various types of disturbances of stream and pond ecosystems were found in the literature. In general, disturbance, perturbation or stress, are defined as an excursion of a system’s state, output or response function (Gerritsen & Patten 1980). In accordance to Niemi et al. (1990), here, the causes of a disturbance are referred as stressors. A stressor might result in a defined disturbance of limited duration or cause long term changes of ecosystem functions. To describe short term or long term perturbation we applied the terms of pulse and press disturbance (Bender et al. 1984), respectively.

We classified two general types of stressors, chemical stress and physical stress. For the analysis of recolonisation processes, constructed wetlands were investigated additionally. Within the group of chemical stress we separated pesticides, other organic chemicals and metals. Whereas pesticides and organic chemicals were used in experimental studies or introduced to aquatic systems by accident, metal pollution, as a result of mining activities, occurs mostly not as short term event and the use of these datasets has to be taken carefully for the given question. Common forms of physical disturbance were all types of manmade disturbance of substrates and communities, as well as flood and drought events. Manmade disturbance include studies conducted in patches of different size to simulate flood events as well as large scale perturbation e.g. resulting from logging activities. Whereas drought was defined as the overall dryness of the sampling site, flood is a dramatic increase in velocity leading to drift of organisms and destruction of habitats. Here, constructed wetlands covered new established streams, ponds or parts of streams as a result of restoration or opencast mining activities. These datasets were analysed separately and compared with each other for testing the ability to use other kinds of stressors for the prediction of the recovery potential of stream and pond ecosystems after pesticide application. We ignored other types of stressors like acidification, structural and saprobial degradation since these are assumed to be press disturbances per se, resulting in long term effects like shifts in community structure, biodiversity and ecosystem functions.

Within an ecotoxicological context, recovery is understood as the return of the community or population structure of a treated system into the range of temporal (in field studies) or spatial (in experiments) controls l. When defining recovery mechanisms a distinction between actual and potential recovery can be made. Actual (or ecological) recovery implies the return of the perturbed measurement endpoint (e.g., species composition, population density, dissolved oxygen concentration) to the window of natural variability in the ecosystem of concern. Potential (or ecotoxicological) recovery is defined as the potential for recovery to occur following dissipation of the stressor to a concentration at which it no longer has adverse toxic effects on the measurement endpoint (Brock and Budde 1994).

Actual recovery can take place either from within the system (autochthonous) or via re-colonization (allochthonous). The internal recovery of populations depends on surviving organisms or on resting propagules not affected by the stressor (e.g., ephippia in Daphnids, individuals being already on the wing in insects). In contrast, external recovery of populations only can occur by migration of individuals from neighbouring systems using active or passive dispersal (Lopez-Mancisdor et al. 2008). As reviewed by Mackay (1992) many studies have shown re-colonization is primarily responsible for restoring a given community under field conditions. Within the allochthonous recovery several mechanisms are available to stream benthos. Wallace (1990) identified four groups of re-colonisation mechanisms in lotic systems, listed by the order of their importance (Williams and Hvnes 1976):

(1) Downstream drift from upstream or tributary areas is by far the most frequently cited mechanism for re-colonisation.

(2) Aerial re-colonization by adults of many insects may represent the primary, if not the sole, source of re-colonization in some situations (e.g. no undisturbed upstream refuges present)

(3) Migration from the deeper hyporheic zone to surface substrates. However, very little is known about the variation of hyporheic faunas within different regions or the influence of local geomorphologies.

(4) A small fraction of amphibiotic macroinvertebrates, mostly holobiotic aquatic species, has been shown to exhibit significant upstream movement. Söderström (1987) suggested that upstream movement is a result of the search for unexploited resources such as food and space, for suitable sites for emergence, pupation or mating or due to the avoidance of unfavourable abiotic conditions and is a compensatory mechanism for offsetting downstream drift.

This ranking might depend on the season as e.g. aerial colonization may be severely curtailed or not existing in winter month (Wallace 1990).

When studying recovery processes, several factors are usually considered such as alteration of food-web interactions and physicochemical parameters, generation time of the affected populations, the supply of propagules or persistence of the chemical, exposure regime and references. Instead of comparing a disturbed system to its pre-disturbance state (temporal reference) one can compare it to a control or an untreated reference site (spatial reference). In this context recovery has taken place when no more significant differences between the disturbed and the undisturbed system can be measured.

Material & Methods

Literature search

A literature search was conducted using the online tools of ISI Web of KnowledgeSM (© Thomson Reuters 2008) and GoogleTM Scholar (© Google 2008). Keywords applied for searches were used alone or in combinations of categories. Four categories, namely (1) stressor, (2) population response, (3) system, and (4) organism, comprising keywords as following:

(1) stressor, pesticide, chemical, herbicide, insecticide, disturbance, flood, drought, impact, constructed, artificial, man-made, and names of several plant protection products

(2) recovery, repopulation, recolonization, colonization, resilience, resistance, succession, dynamic

(3) pond, lake, stream, creek, river, freshwater, wetland, microcosm, mesocosm, lentic, lotic

(4) algae, periphyton, macrophyte, invertebrate, macroinvertebrate, insect, fish

Furthermore, cross references of collected articles, either literature cited or related-article functions of online tools (e.g. ScienceDirect®, © Elsevier B.V. 2009), allowed subsequently conducted searches. Case studies including at least one endpoint of recovery were selected on the base of four criteria:

(1) Description of the system or site characteristics available

(2) Disturbance caused by a stressor which is described clearly, including type of stressor, a measure of quantification as well as spatial and temporal dimension

(3) Description and quantification of a pronounced effect related to stressor described

(4) Data on recovery processes are available, including pre-disturbance or reference data, or data indicating stable, post-disturbance populations

The isi web of knowledge database provided a list of 2348 and >100000 citations when searching for the topics ("recolonization") or ("recovery") respectively. Since ("recovery") includes recovery of ecosystems as well as recovery in terms of analytical chemistry, combinations of keywords were applied as described above. For example, combining the topics ("recovery" AND ["stream" OR "pond"]) resulted in a list 16245 citiations, whereas the combination of the topics (["microcosm" OR "mesocosm"] and ["recovery" OR "recolonisation"]) led to 274 results.

Prossessing of case studies

All studies were reviewed to gain information about the system referred to, study processing, the stressor and the resulting effect, recovery endpoints observed and the organisms considered. In order to compare the cases selected, information on the system like the region and country, the watershed or surrounding landscape, classification of the type of the ecosystem, and a short description of the waterbody and velocity were included in a data base as well as information on the study like sampling method, season in which the stressor occurred, the year of the study and the year of the publication.

The type of the ecosystem studied could first be differentiated into ‘real’ ecosystems in the field, e.g. streams, ditches, ponds in a landscape which were monitored with respect to the effect of and recovery from a disturbance, which was either a given disturbance, (e.g. timber logging or mining activity), or a manipulation (e.g. insecticide application). References for effects and recovery found in theses cases were the state of the disturbed system before disturbance, the state of a similar, but undisturbed system, or a theoretically derived reference state. With respect to streams, duration and magnitude of a given disturbance may vary with stream characteristics as reviewed by Wallace (1990). The systems included in the current review incorporated all kind of running waters differing with size and velocity (ditches, steams, creeks, and rivers), substrate, geographical position or altitude.

The other class of studies comprises model ecosystem studies in micro- or mesocosms, enclosure systems, or artificial streams. In contrast to the field studies, these tests offer replicated control and treatments. At least in theory, here the chemical is the only stressor differating the test units. Depending on the study desingn, the size of the model ecostystem, its homogeneity and their spatial isolation can result in worst case conditions regarding effects and recovery, i.e. due to reduced potential for re-colonisation. Running systems constructed for experimental purposes are referred to as artificial streams. The lotic systems which are of the main interest in the project’s framework can be described as small, possibly intermittent streams or ditches, with low velocity and rich riverine vegetation.

Recovery endpoints refer to any value of biological measures commonly used for quantifying aquatic communities classified as abundance, biomass, taxa richness, diversity, community composition (e.g. index of similarity or principle response curves), indicator organisms or first occurrence. Usually values of biological measures were compared to reference states or systems to estimate recovery, here grouped into the categories state control (spatial reference) state pre-treatment, or comparison post-treatment (time temporal reference). In case of pulse disturbances, recovery time to various endpoints was estimated from the time a stressor occurs in the system. Time to recovery from press disturbances was estimated from time when exposure ends, e.g. in case of restoration success after chronic metal pollution. In order to avoid miss-interpretation, disputable results were reviewed by at least two of the authors. In case of endpoints with recovery time shorter than sampling intervals time to recovery was recorded as “< date of next sampling” values, in cases where recovery did not occur during study period, time to recovery was indicated as “> Study period”.

Stressors were classified and the resulting disturbance shortly described including a measure of quantification, spatial dimension (ponds and mesocosm [ha], (artificial) streams [km]), and if available the duration of exposure [d] (if not 1 hour was assumed). The effect caused by a stressor was shortly described, the presence of secondary effects (yes/no) and the effect size noted as percent reduction of the endpoint referred to, as well as the duration of effect [d] if available.

For data analysis organisms were first grouped into zooplankton, macrophytes, algae, macroinvertebrate, and fish. and then further subdivided based on taxonomy (e.g. order in case of insect macroinvertebrates). The lowest taxonomic level available in studies was also listed. The data collected was organised in a data base. The information available for each case, as described above, is listed for each taxon and is referred to as a database entry.

Time to recovery data of selected taxonomic groups was plotted as box-whisker-plots, including the calculation of median and quartiles, using SigmaPot 8.0. Cumulative frequency of taxonomic data [%] was plotted as a function of time to recovery for lentic and lotic systems. Analyses include all database entries with observed time to recovery as well as entries without any recovery observed in case the duration of the original study was ≥ 1 year. Entries of studies < 1 year and not containing recovery data were excluded from further analysis. Probit analyses were conducted using the statistical package of ToxRatPro.

Results

In the following chapter, data on recovery processes and time to recovery is summarized regarding different taxonomical groups.

Overview on the case studies covered in the review

On the base of title and abstract, a total of 471 and 152 publications were collected for lotic and lentic systems respectively. Applying the criteria as described above, 397 publications were rejected. At least, the selection included 150 articles in lotic systems and 76 articles in lentic systems, resulting in a total number of 148 cases studies and 908 database entries. In comparison, in a comprehensive review of case studies Niemi et al. (1990) identified 164 aquatic systems with at least one endpoint of recovery measured. Based on their findings, the emphasis of our literature search was drawn to studies published since 1990. Studies published in the period 1990-2008 account for 65% and 86% of the cases included in this review in lotic and lentic systems respectively.

Of the case studies included in this review, 63% were lotic, providing 54% of the data available for recovery endpoints, and 37% lentic accounting for 46% of the entries. The aquatic systems originally studied were mostly distributed across North America (54%) and Europe (28%). Only few cases were found for Australia/Oceania, Africa, Asia, and South America. The duration of the original studies varied with the objective and the type of stressor referred to. Studies were predominantly conducted within a period of 1-3 years (41%), or shorter (40%), while longer studies were found less frequently. In most studies investigation started or a stressor occurred in summer (41%) or spring (16%) whereas in 21% no information on the season was available. The most common stressors (Fig. 1) found were the groups of chemical (n=76) and physical stressors (n=23) whereas data on flood (n=14), drought (n=17) and constructed aquatic systems (n=17) were less common, probably due to our search algorithm.

Of the 908 database entries listed, the biological measures most widespread used were the total abundance of organisms (n=408) and the abundance of single taxa (n=278). Furthermore taxa richness (n=93) and the biomass of taxonomic groups (n=74) were used in a number of studies to describe recovery processes. Community measures namely community composition (n=33), diversity (n=21), and indicator organisms (n=1) were used less frequently (see also fig. 2).

When attempts were made to describe recovery processes or to quantify times to recovery, reference states, in a spatial or temporal manner, were used as a benchmark. Spatial references, e.g. unstressed upstream reaches and tributaries or neighbour ponds, were used in most entries (n=372) whereas the availability of pre-disturbance data allowed a set-actual comparison in 233 database entries. With respect to the lack of pre-disturbance data in a number of entries (n=294) recovery was assumed when an endpoint reached stable post-treatment values. In few endpoints (n=9) no information was available on the reference used. In 19% of the database entries no recovery was found during the study period.

In order to quantify disturbance scenarios, effects were commonly described as percent reduction of endpoints In 55% of the database entries effect was > 90% and less than 50% in 3% of all entries. Secondary effects, e.g. algae growth due to ceasing of grazers, were reported in 32 studies.

The majority of the database entries were identified for macroinvertebrates (n=629, Fig. 3). In comparison to Niemi et al. (1990) data for zooplankton (n=133), algae (n=50), and aquatic macrophytes (n=51) increased in the recent years. In this study 45 entries for fish were included. Within the group of macroinvertebrates most of the data was available for Diptera (n=114), Ephemeroptera (n=64), Coleoptera (n=43), Trichoptera (n=42), and Heteroptera (n=32). In a number of studies entries were clustered in functionals groups (n=221) including feeding groups as well as total abundance and biomass of macroinvertebrate. Data was sparse for other macroinvertebrates like crustaceans and molluscs.

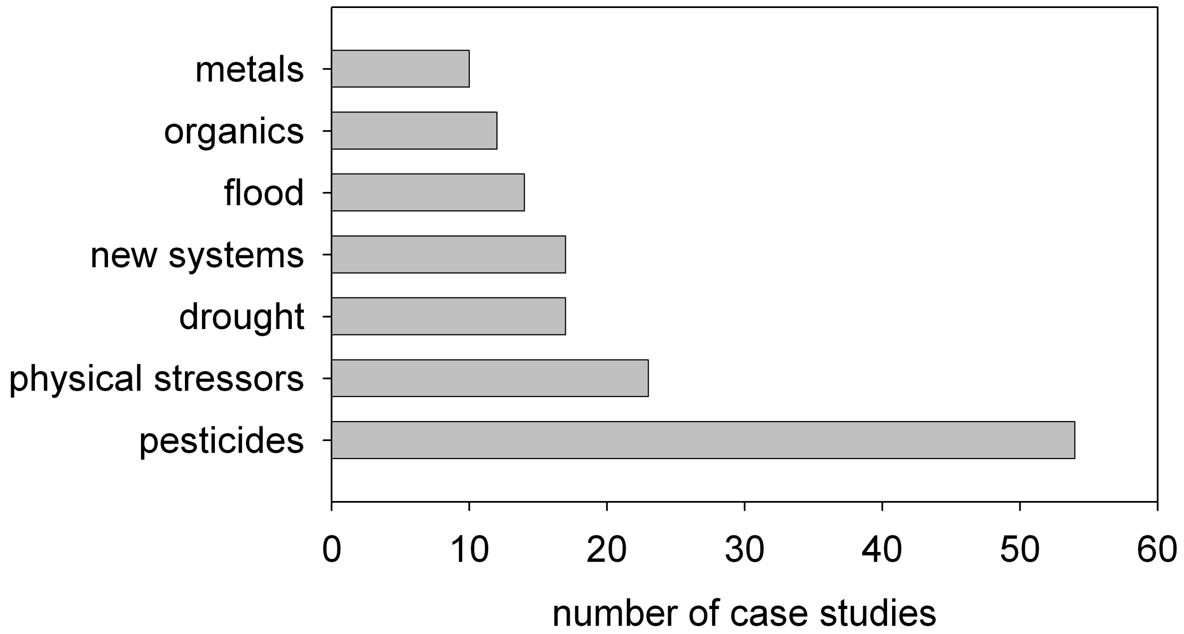

Fig. 1: Number of cases in five categories of stressors included in the study

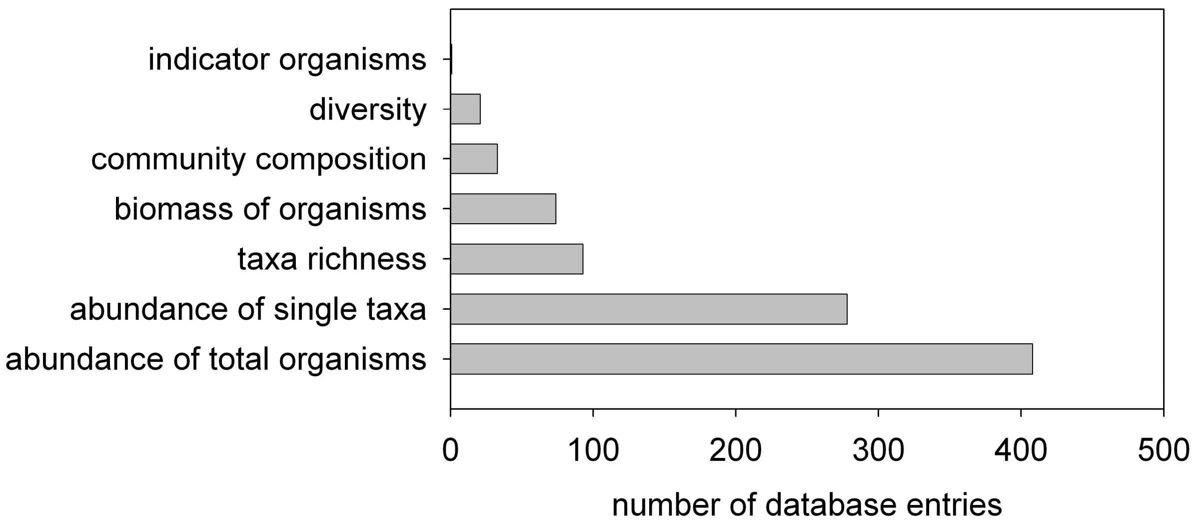

Fig. 2: Number of database entries graded in seven categories

Overview on selected taxonomic groups

The case studies extracted from the literature were analysed for recovery times and recovery mechanisms after disturbance. Results are presented for the groups algae (phytoplankton and periphyton), macrophytes, zooplankton, macroinvertebtrates and fish.

Time to recovery data is plotted for each taxonomic group, ordered by their outer quartile in lotic and lentic systems (Fig 4 & 5). From these figures a series of recovery or recolonisation can be derived. Lentic zooplankton and phytoplankton recovered in nearly all cases within one year. Fastest recovery was observed in algae followed by Ephemeroptera and Diptera independent of ecosystem type. Lowest recovery potentials were found for Trichoptera, Odonata, Plecoptera and Crustacea in lotic systems. In lentic systems benthic crustaceans, Trichoptera, Mollusca and Coleoptera did mostly not recover within one year. If the analysis is restricted to pesticides the low amount of available data does not allow any interpretation of recovery for specific taxonomic groups, but the overall pattern seen in the whole dataset is mostly reflected.

This dataset is analysed in more detail in the following chapters for selected taxonomic groups.

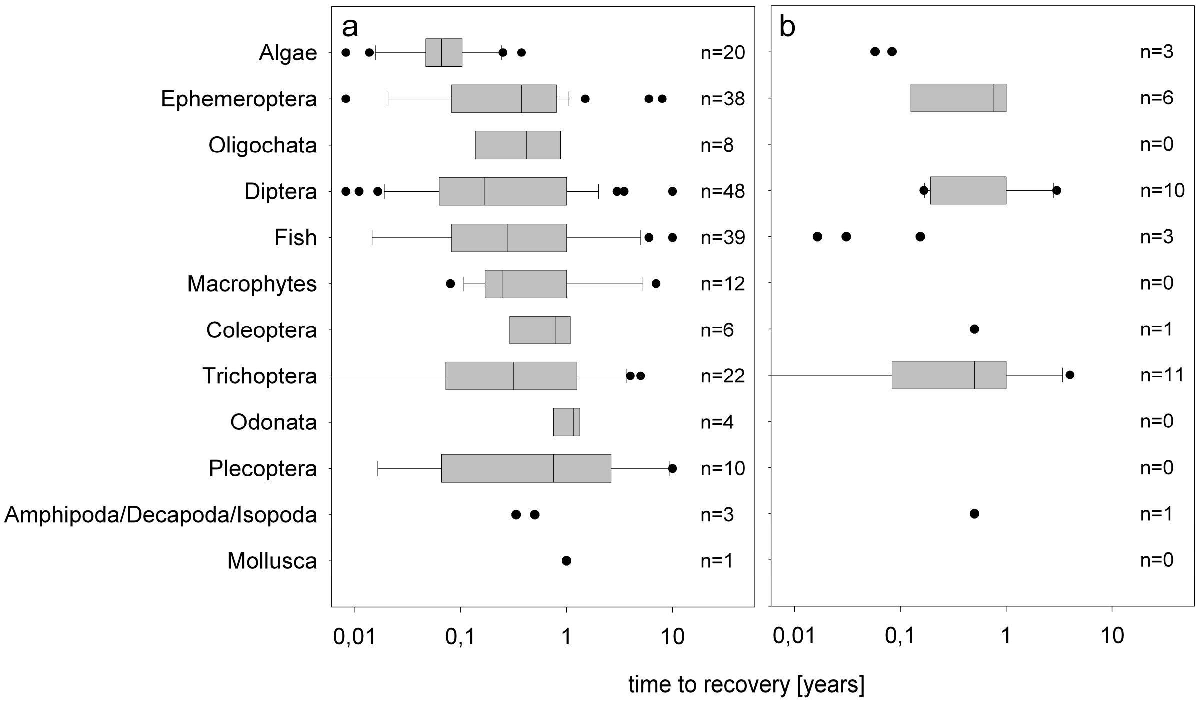

Fig. 4: Time to recovery of selected taxonomic groups in lotic systems after stress: (a) all stressors included in the study, (b) pesticide. Boxes represent median, inner and outer quartile. Taxonomic groups sorted in descending order by their outer quartile.

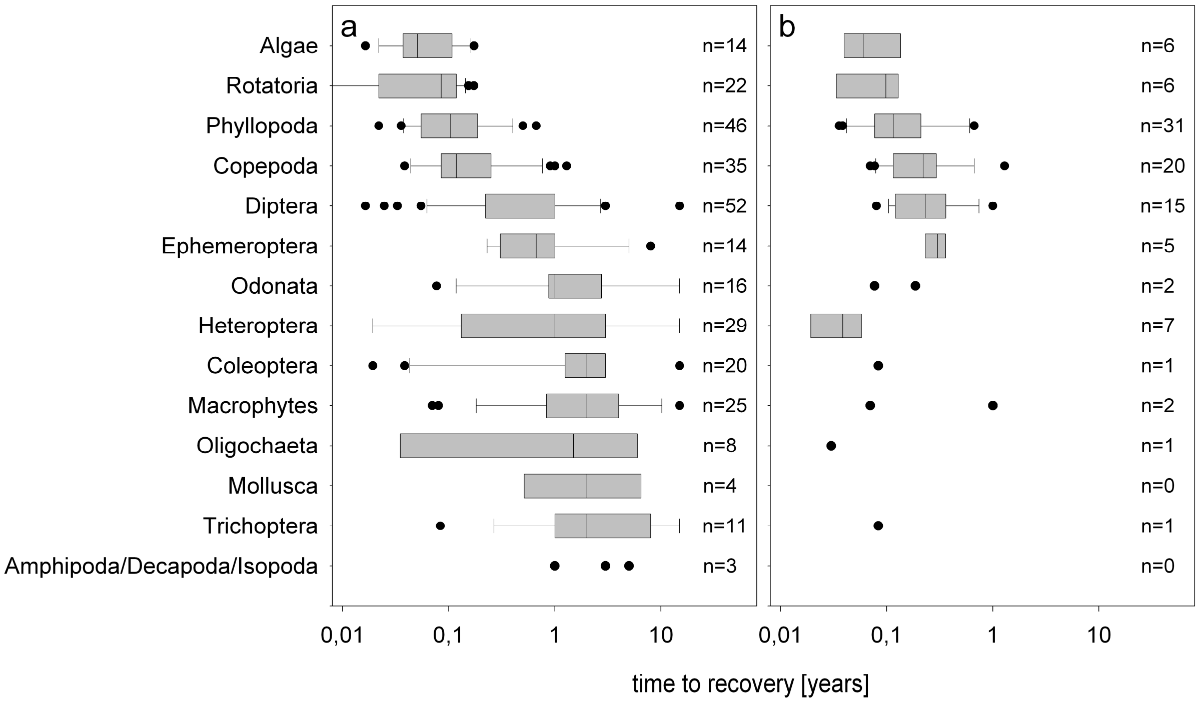

Fig. 5: Time to recovery of selected taxonomic groups in lentic systems after stress: (a) all stressors included in the study, (b) pesticide. Boxes represent median, inner and outer quartile. Taxonomic groups sorted in descending order by their outer quartile.

Algae

Algae could be differentiated into phytoplankton, periphyton and macroalgae, for example *Chara,* which are discussed here in the section on macrophytes. Overall 12 case studies for lotic systems (periphyton) and 10 case studies for lentic systems (phytoplankton and periphyton) were found.The median times to recovery were calculated as 0.07 and 0.06 years for lotic and lentic systems respectively. The maximum time to recovery for algae was found at 0.4 years. Furthermore, one study conducted in Alaska did not show any recovery after an oil spill within 6 years due to the very low temperature . However, in most studies, except mesocosm, community endpoints, like algae biomass or chlorophyll, were used. Therefore no conclusion for the recovery of single species can be drawn. Effects on and recovery of phytoplankton is often studied in micro- and mesocosm studies (including enclosure systems). As to be expected, the available data from model ecosystem studies indicates a high potential of recovery due to the relatively high growth rates (compared to the other groups considered here) and their usually high numbers which makes total exctinction unlikely.

Considering, the intrinsic growth rates of most phytoplankton species, their broad distribution, the dispersal potential due to passive transport and the redundancy of the phytoplankton as a functional group, recovery of phytoplankton can be expected even after large reductions in numbers within a few weeks if the environmental conditions (temperature, light, nutrients) are favorable. Thus, recovery of phytoplankton is not considered as a critical topic with respect to pesticide effects in the field

Periphyton has been less intensively studied in lentic micro- and mesocosm studies than phytoplankton. However, in lotic systems where periphyton can be the most important primary producer, 20 database entries provide data on periphyton recovery. The recovery time of periphyton communities exposed to heavy metals may take longer than for the other types of disturbances discussed (Steinman & McIntire 1990). Finally, community structure may affect periphyton recovery time. Kaufman (1982) noted that older periphyton communities had a lower resistance than younger communites. This is analgous to the slower recovery times reported for ungrazed communities relative to grazed communities after a flood event, in that older communities are more complex and dynamically fragile (May 1975). However, good dispersal abilites, high production rates, short generation times and flexible life history strategies should enable periphyton biomasss to recover rapidly after a disturbance event in most lotic ecosystems (Steinman & McIntire 1990).

Macrophytes

Effects on macrophytes (including macroalgae like e.g. *Chara*) may be differentiated in a temporary inhibition of growth, e.g. by photosynthesis inhibitors or inhibitor of fat acid metabolism, and destruction of plants or part of the plants. For the first type of effects, the individual plants might be able to recover by reaching usual growth rates again but recovery to the standig crop of unexposed plants is not possible until growth is limited by e.g. nutrient depletion or low light or temperature. It is still a matter of debate if a such a recovery of growth rates can be accepted assuming that the period of growth inhibition is sufficiently short to prevent indirect effects (e.g. on macrophyte grazers).

If parts of the plants or the total plants are destroyed, recovery depends on e.g. the ability to grow again from the surviving roots or shoots, the growth rate and the dispersal potential. Thus, recovery in the field is likely not a problem for small floating plants with high growth rates, i.e. *Lemna*, or other common plants which might be 'pests' like *Elodea spec.*

For the assessment of the recovery potential of aquatic macrophytes agricultural landscapes it has also to be considered that ditches and streams are often managed in regular intervals to maintain their hydrological function. Thus, the macrophyte community in these water bodies is adapted to the periodic dredging by selection of relatively fast growing species and/or species able to grow from remaining roots or pieces of shoots. In addition, seasonal die-back of several macrophyte species (e.g. *Potamogeton natans*) during winter with complete new development from the roots in spring, has to be considerd for assessment of stressor effects, especially in autumn.

Only few case studies were found where recovery of macrophytes was analysed. In most field studies found, the focus was on macroinvertebrates. In the past, macrophytes (except for *Lemna* as a standard test species) were not very often studied for pesticide risk assessment, even in mesocosm studies with herbicides.

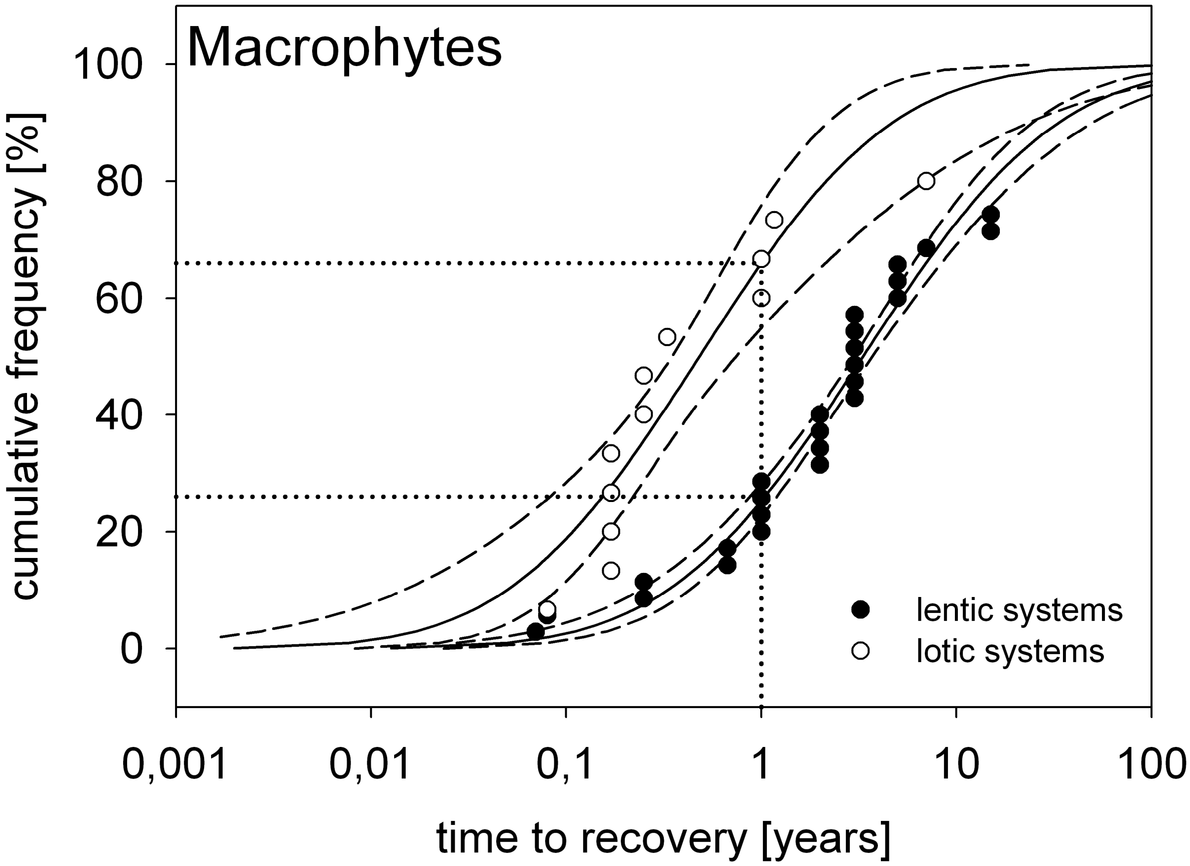

Fig. 5a: Cumulative frequency distributions of observed time to recovery of macrophytes after disturbance in lotic and lentic systems. Dots represent data derived from the literature, lines represent regressions of probit analyses in lotic (r^2=0,782) and lentic (r^2=0,913) systems (solid: mean, dashed: 95% confidence intervals). Dotted lines indicate the percent of endpoints where recovery was found within one year.

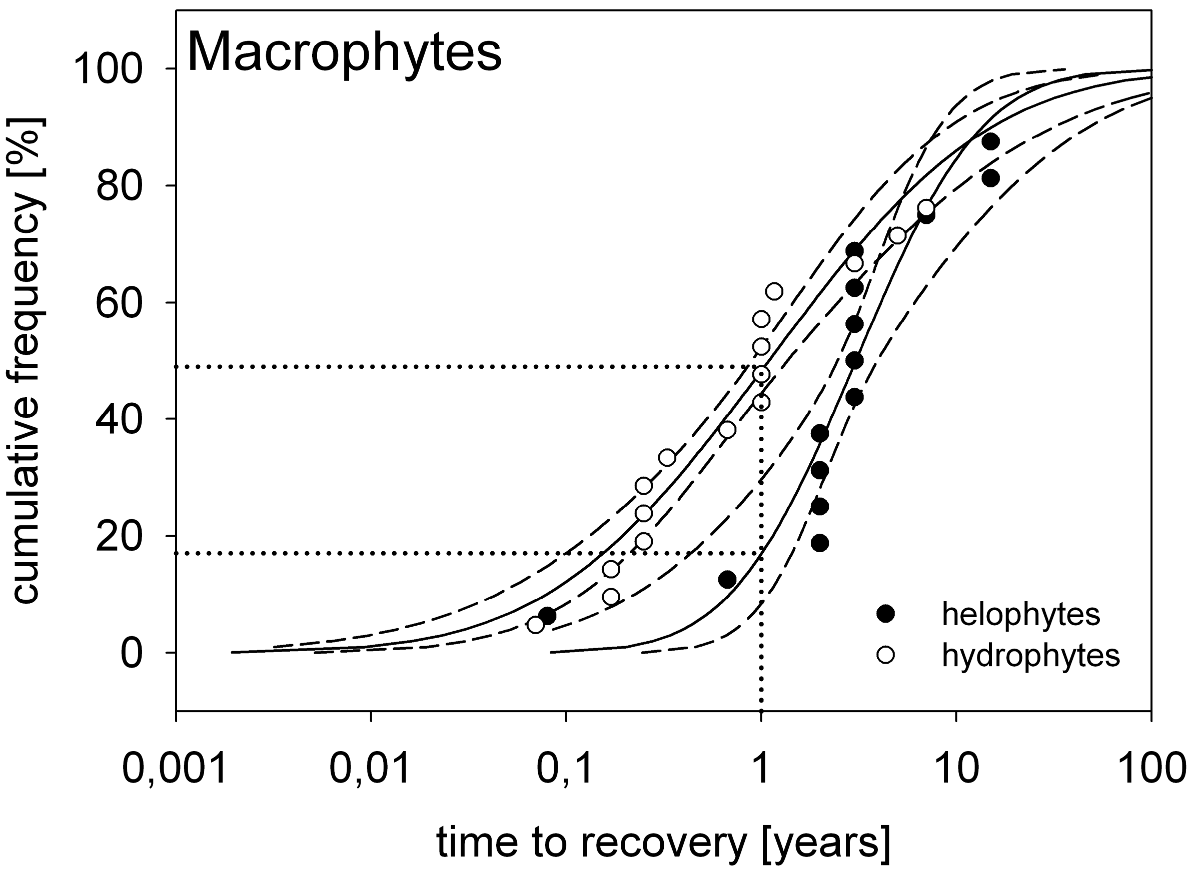

Fig. 5b: Cumulative frequency distributions of observed time to recovery of hydrophytes and helopyhtes. Dots represent data derived from the literature, lines represent regressions of probit analyses in hydrophytes (r^2=0,900) and helophytes (r^2=0,675); solid: mean, dashed: 95% confidence intervals. Dotted lines indicate the percent of endpoints where recovery was found within one year.

Zooplankton

Zooplankton is often the main consumer group studied in micro- and mesocosm studies and therefore a large data base should be available to analyse the recovery potential of different species. However, usually the focus of the publications is on the determination of effect thresholds or on recovery within the study period. Papers with an explicit evaluation of the relationship between the magnitude of the effect and the resulting time for recovery back to the range of the controls are rare. One exception is the publication of O'Halloran et al. (1999) on effect of Nonylphenol on zooplankton where a table of taxon related recovery times and the magnitude of effects can be found. For most taxa, recovery from effects on abundance of 90 % or more was observed within a few weeks. Exceptions were the cladoceran species *Acroperus* and calanoid copepods where no recovery within 9 weeks after end of exposure was given.

This is in line with findings from other model ecosystem studies and field surveys and can be explained by the life cycle characteristics of the different zooplankton taxa. Most rotifers, cladocerans and copepods are characterized by large intrinsic growth rates (see e.g. Barnthouse 2004) and several species have resting stages, e.g. ephippia for *Daphnia*, resting eggs of rotifers.

However, also within the zooplankton r- and K- strategists can be differentiated. Calanoida are an example of a zooplankton taxa with relatively long generation times.

In addition to their life cycle properties the pure number of zooplankton individuals might enhance recovery by the high probability that some individuals can survive a disturbance.

With respect to allochthonous recovery, the dispersal and thus the re-colonisation potential is limited to passive transport of the animals or resting stages via water, wind, water fowl, larger invertebrates (e.g. Notonecta,) or other animals. However, data on the colonization of new ponds indicate that this dispersal can be very effective.

Leeuwangh et al (1994) compared the effects of the insecticide Chlorpyrifos in modelecosystems of different complexity. They found that the recovery started when concentrations had fallen down to the range of the EC10 derived in acute single species tests. Thus after reduction dowin to less then 10 % populatons of D. magna were able to recover within 3 weeks after the start of the exposure back to the level of the controls in an algae - Daphnia test system. In 600 L water sediment microcosms including several species of algae, zooplankton and macroinvertebrates, phyllopoda and copepoda were able to recover from resting stages like ephippia within a few weeks while larger crustaceans like Gammarus and Asellus as well as insects like Chaoborus and Cloeon could not recover due to the lack of resting stages and the prevention of recolonisation under the experimental conditions. "In the case of chlorpyrifos, the onset of recovery of the cladocerans starts at an approximate concentration corresponding to the NOECacute/chronic or EC10(48 h).Unlike the indoor derived microcosms, where the isolated situation and the experimental conditions hindered recolonisation, the mesocosm experiments showed recovery of insect species. Although the onset of recovery could not be precisely determined, the observations reflect the determinant role of the insect life-cycle in the rate of recovery." Leeuwangh et al. 1994)

Usually the test item is applied uniformly to the whole test item, either by application onto the whole water surface or by introduction directly into the water column eventually followed by gently stirring. Thus, compared to the field siturtion less refuges of low or absent exposure are expected which result in a less favourable situation for autochtonous recovery. Recently Lopez-Mancisidor et al. (2007) conducted a study explicitely to invetigate the effects of refuges on effects on and recovery of zooplankton exposed to an insecticide. Therefore, they used experimental outdoor ditches of 60 m³ where the fast dissipating insecticide lufenuron was sprayed on 0, 33, 66 or 100 % of the water surface.

In conclusion, recovery from short-term exposure is possible for most zooplankton taxa within a few weeks. Exceptions are univoltine species, e.g. calanoidae. However, recovery also for these taxa can be expected until the next season in most cases due to reproduction of survivors, resting stages and or re-colonisation.

Macroinvertebrates

Macroinvertebrates are a diverse group including arthropods, mollusks and oligochaetes Various feeding types are included ranging from primary consumer to predator species. Some macroinvertebrates are able to disperse via flight in adult stages, whereas larvae of these groups and other wingless taxonomic groups are restricted to passive dispersal or in-stream-movement. In the following, the expected time to recovery of different macroinvertebrate taxa will be discussed.

Heteroptera

Recovery or colonisation of Heteroptera was observed in 8 lentic cases and 1 lotic system where bugs occurred in pools shortly after establishing a river (Malmqvist et al. 1991). Of the 30 heteropteran database entries included in the current review most were observed for

Notonectidae (50%) and Corixidae (20%). Heteroptera, in particular Notonectidae and Corixidae, are generally known as fast colonisers due to their high capability of flight (Barnes 1983, Solimini et al. 2003), leading to a median time to recovery of one year in lentic systems. Even faster recovery of backswimmers was found in sahelian desert ponds after insecticide treatments (Lahr 2000). *Anisops* adults reappeared in the systems a few days after treatment. As a result of the immigration of flying specimens the time to recovery of nymphal stages was less than or equal to 3 weeks. The *Anispos* species inhabiting temporal desert ponds are referred to as vigorous flyers and colonists of water impoundments in the beginning of the rainy season.

However, slow recovery of aquatic Heteroptera was found in two cases. One study observed first occurrence and afterwards stable biomasses of Heteroptera five years after flooding a shallow artificial lake in northern Sweden (Danell & Sjöberg 1982). In a second study the succession of different aged ponds, a result from opencast clay mining, was observed. Besides the occurrence of early colonising bugs in this study, three species of Corixidae, did not appear in constructed ponds which were younger than four years of age. Late colonisers were *Sigara scotti*, *S. falleni* and *S. distincta*, which are characteristic for permanent waters (Barnes 1983). No recovery of aquatic Heteroptera was found in one case investigating the colonisation of newly build bog ponds (Mazerolle 2006). In the study, Belostomatidae and Nepidae already colonised manmade pools but captures of bugs were fewer than in natural pools four years after creation.

Coleoptera

A total of 16 case studies and 43 data base entries were identified in aquatic Coleoptera, of which 38% of the sytems and 77% of the entries were lentic. The median time to recovery was 2 and 0.56 years in lentic and lotic systems respectively. Like Heteroptera aquatic beetles are among the earliest colonists of new freshwater habitats (Fernando 1958, Barnes 1983). In particular, Fairchild (2000) reported the occurrence of Hydradephaga in early successional stages whereas hydrophiloid beetles occurred in later stages of the lentic ecosystems. Fast recovery after insecticide treatment was reported in highly mobile Gyrinidae (Fairchild 1993) as well as the colonisation of a new pond within one year after creation by Dyticidae and one species of Hydrophiloidea (McDonald & Buchanan 1981). No recovery of both Hydadephaga and Hydrophiloidea was observed in one lentic case where bog specialists already colonised constructed ponds four years after creation but did not reach abundances of natural control systems (Mazerolle 2006).

In 72% of the lotic entries (Fig. 6) recovery was reported within one year after disturbance by pesticide, flood and drought (Scimgeour et al. 1988, Paltridge et al 1997, Liess & Schulz 1999, Collier & Quinn 2003) or the construction of a new stream (Malmqvist et al 1991). A longer time to recovery was found in a subtropical fourth order stream affected by an crude oil spill. In this study, a *Hydroporus* species first reappeared 26 month after the spill (Harrel 1985). No recovery within the study period was found in two cases. In one case a single species, *Heterlimnius corpulentus*, was most severely affected by a flash flood and recovery was poor due to low dispersal rates (Molles 1985). In the Breitenbach, Germany, the flightless, long living beetle *Oulimnius tuberculatus* was still rare 6 years after an accidental entry of Cypermethrin (Zwick 1992).

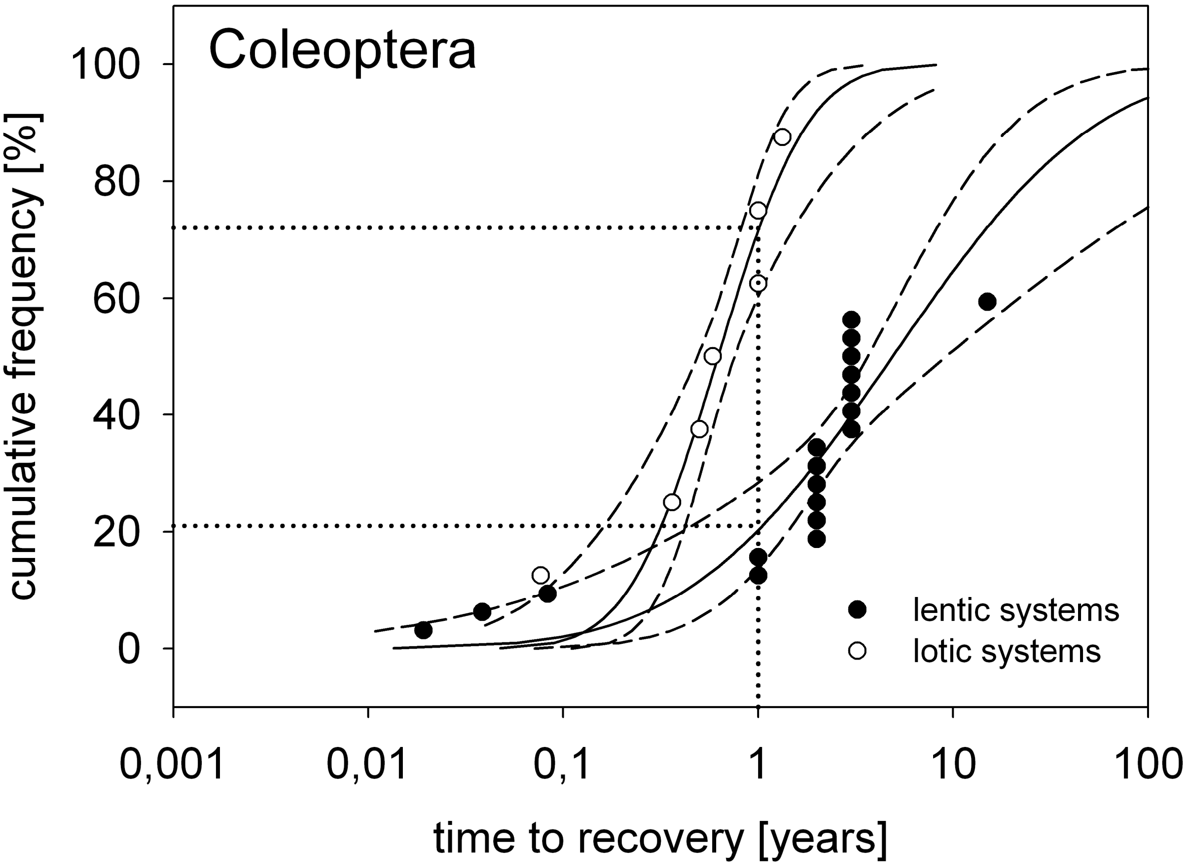

Fig. 6: Cumulative frequency distributions of observed time to recovery of Coleoptera after disturbance in lotic and lentic systems. Dots represent data derived from the literature, lines represent regressions of probit analyses in lotic (r^2=0,867) and lentic (r^2=0,554) systems. Legend: see above

Oligochaeta

For oligochaeta 9 lotic and 5 lentic case studies were found, the median time to recovery were 0.42 and 1 year respectively. In two cases no recovery was observed within the study period. *Tubifex tubifex* was not able to recover to the pretreatment state within one year after multiple insecticide runoff events (Liess & Schulz 1999). The other example was found for pollution of an artificial stream with selenium (Swift 2002), In this study *Tubifex spec*. again was not able to recover within a 3 year period. For five studies (Miller 2006, Barnes 1983, Koskenniemi 1994) investigating recolonization of constructed wetlands only slow recolonisation was found for oligochaetes. The recovery to maximum abundance after the construction of new wetlands was between 2 and 15 years. Very fast recovery within half a year was found in case studies investigating drought (Otermin 2002, Fuller 2985, Harriman 1988) or physical disturbance for a small scale in which the oligochaets were able to migration from hyporheic zone or neighbor patches (Otermin 2002, Fuller 1985). No differences in time to recovery of oligochaete endpoints were found between lotic and lentic systems (fig. 7).

Overall it seems that oligochaetes miss a good dispersal potential and therefore recolonization is slower than for other species and fast recovery found were due to unaffected species within the sediment or due to intrinsic recovery.

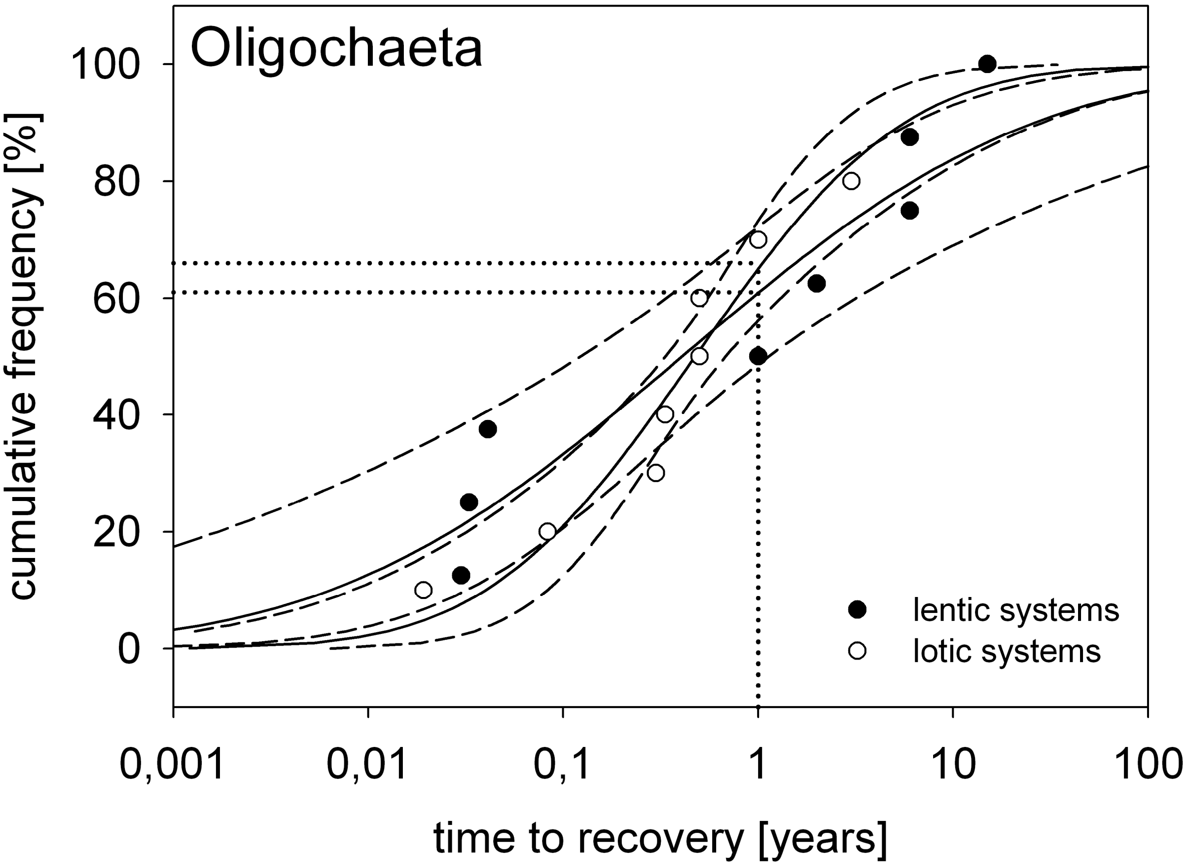

Fig. 7: Cumulative frequency distributions of observed time to recovery of Oligochaeta after disturbance in lotic and lentic systems. Dots represent data derived from the literature, lines represent regressions of probit analyses in lotic (r^2=0,871) and lentic (r^2=0,838) systems. Legend: see above

Odonata

Odonata were investigated in 3 lotic and 9 lentic case studies with median time to recovery of 1.17 and 1 year respectively. No recovery was found within 4 years in one case study investigating the colonization of a new pond (Mazerolle 2006). In another case study investigating the colonization of a new pond too, recovery time from 1 to 15 years were found for different odonata species (Barnes 1983), whereby *Lestes sponsa* and *Orthetrum cancellatum* showed the lowest colonization potential with a maximum population abundance 15 years after the treatment. A third study investigating colonization of a new pond found the highest population abundance after 8 years (Danell 1982). Further five studies investigating the colonization of new constructed ponds and streams found times to recovery from 0.13 to 1 year (Malmqvist 1991, Barnes 1983, McDonald 1981, Solimini 2003, Christman 1993). For an oil spill in a stream recovery times of 1.3 years were found (Harrel 1985). For insecticide treatment two case studies were found for ponds in which the time to recovery were 0.08 and 0.19 years (Fairchild 1990, Sundaram 1991) and one mesocosm in which no recovery was observed within the study period of 70 days (Ward 1995).

Overall dragonflies seem to have a low potential for recolonization, but it has to be taken into account that they are predators needing a stable sufficient supply of prey. This becomes obvious by comparing recovery in new constructed ponds, where prey has to develop first, and other stressors like insecticide treatment where insensitive species may remain as food resource, with time to recovery of 7 and 0.1 years respectively.

Mollusca

For mollusca only 3 lotic and 4 lentic case studies were found, with 3 and 6 data base entries respectively. For either case evaluated an effect of 100% was observed, since the case studies dealed with completely new constructed ecosystems or the disturbance resulted in a total disappearance of snails and mussels.

For Savannah River, a recovery time for *Physa* spec of one year was found after a contamination with coal ash that was associated with low pH and included e.g. Sulphates, Arsenic, Copper, Selenium and Zinc (Cherry 1979). After a crude oil spill in Turkey Creek it took even 26 month for *Sphaerium* spec to reappear. Although, the decrease in oil concentration resulted in a gradual recovery of several taxa, no clean water taxa were collected and complete recovery had not occurred 26 months after the spill when the study was terminated (Harrel 1985). Within the third lotic case study, colonization of constructed gravel bars in Tombigée River was investigated (Miller 2006, McClure 1985, Bingham 1989). For invertebrates the colonization of the newly placed gravel shoals was rapid, since within stream movement was the main recovery pathway expected, and the mussels *Obliquaria reflex* as well as *Corbicula fluminea* occurred immediately after construction.

Comparing the lentic cases, recovery rates between seven days and more than six years were found. Actually, within a new constructed shallow lake (Kyrkosjarvi Reservoir, Finland), *Pisidium* spec and *Sphaerium corneum* did not reach abundances alike comparable ecosystems within the study period of nine years. Thus, recovery was not completed for molluscs by study termination (Koskenniemi 1994). In contrast, the gastropod *Nemalia* spec reappeared very fast after drying and liming of Ebrié lagoon pond. Here, a recovery rate of seven days was found (Guiral 1994). Within the investigation of different aged man made Purbeck ponds *Lymnea peregra* and *Acroloxus lacustris* occurred after 2 years (Barnes 1983) and for the new shallow Veittijärvi lake, it took more than six years until mollusc biomass was recovered (Danell 1982).

In conclusion, molluscs seem to recover faster if immigration from directly connected neighbouring areas is possible. Since they have no terrestrial life stages, for the colonization of new ecosystems they are dependent on passive dispersal, which may result in recovery rates of several years.

Ephemeroptera

Recovery processes for the larval stages of Ephemeroptera were investigated in 26 lotic and 12 lentic case studies with 42 and 20 database entries respectively. Most database entries observed for mayflies in lotic systems were for Baetidae (40%). Other abundant lotic families were not included this frequent (e.g. Leptobhlebiidae 8, Heptageniidae 2, Ephemerellidae 1 database entries). Since especially Baetidae are known to be prominent in drift and good swimmers, they are often found among the first arrivers after a disturbance in lotic systems (Mackay 1992). In general, for mayflies in streams early recovery within one year by drift and recolonization from upstream or nearby sources was found, leading to a median recovery time of 0.26 years.

Especially for Baetidae in streams, several case studies with very fast recovery within only a few days were found (Dosdall 1989, Brooks 1991, Paltridge 1997, Tikkanen 1994). As some Baetidae seem to be able to survive drought in the hyporheos and venture fast after the first flow, Paltridge (1997) found reappearence of Cloeon only 3 days after the end of a six month drought by migration from the hyporheic zone in Magela Creek. For experimental small scale experiments Doeg (1989) found recovery rates for Baetis and Leptoblebiidae between 8 and 71 days depending on the season after physical disturbance in the Acheron River, Australia. In this case, stones were removed out of the stream, completely cleaned and defaunated and introduced again. The time to recovery of abundance compared to a control reach could directly be linked to the size of the patch to be colonized again and the distance between this patch and the source of colonization. Furthermore, the same observations were made by Brooks (1991). They found fast colonization of mayflies by surface movement from neighbouring intact areas after experimental defaunation of small stream bed plots.

Regarding disturbances at a larger scale, recovery times for mayflies varied from only a few days up to several months. For example, in North Saskatchewan River, Dosdall (1989) observed recovery of *Baetis tricaudatus* to pre-treatment state by drift from upper undisturbed reaches only 5 days after a Methoxychlor application in spring that reduced total abundances between 45 and 99%. Specht (1984) described recovery of mayfly abundances after a two years heavy metal discharge resulting in a decline of mayfly numbers from 40% to 20% of all insects within 5 to 7 month depending on family compared to an upstream reach. And for Sonadora river in Puerto Rico, recovery of mayfly abundances after an organic disturbance by chlorine-bleech took 3-4 month (Greathouse 2005).

Within the dataset, only few case studies with recovery times more than one year were found. In McCoy Branch, abundances of Baetis recovered within 1.5 years after a long-time heavy metal discharge for nearly 30 years. Nevertheless, taxa richness of mayflies was not recovered completely by study termination after six years (Smith 2003, Ryon 1992,1996). Another case study with very long colonization periods of mayflies, were the constructed wetlands in Glacier Bay that were fed by meltwater (Milner 1987, 1999, 2000,Flory 1999a, b). Although mayflies were one of the first invertebrates to colonize here, it took eight years to achieve the state of the control reach, due to overall low water temperatures.

Compared to lotic systems, recolonization of mayflies in lentic systems mainly was driven by allochthonous recovery and a median recovery time of 0.5 years was calculated. For new constucted shallow lakes and experimental ponds colonization periods for lentic ephemeropteran species within less than one year were observed by Christman (1992) and Koskenniemi (1994). In other cases colonization took more than one year (Barnes 1983, Danell 1982). Since colonization processes are mainly driven by dispersal of adults, populations will develop faster if unaffected neighbouring ponds are available and if the colonization period coincides with reproductive periods of the regarded mayflies.

Besides the new constructed lakes and ponds mayflies were also investigated in mesocosm experiments with application of pesticides. For these case studies fast recovery as well as no recovery within a given study period could be observed depending on species and test design. Sundaram (1991) found recovery of Caenis abundance within 110 days after a significant short-term effect due to application of diflubenzuron. After application of deltamethrin in mesocosms, species of the families Baetidae and Caenidae showed complete recovery within 149 days latest if recolonization from neighbouring ponds was allowed. Whereas in cases only autochthonous recovery was investigated no recovery could be observed (Caquet et al. 2007). Species dependent recovery was found by van den Brink et al. (1996) after application of chlorpyrifos in experimental ditches. While Cloeon recovered after 84 days latest, Caenis did not recover at all within the study period of one year. When chlorpyrifos was used for indoor microcosm experiments with allochthonous recovery excluded no recovery of ephemeropteran species could be observed at all (Leeuwanght et al. 1994).

In conclusion, mayflies seem to be good and early colonizers of disturbed reaches as long as needed habitats are available. They recover faster in case disturbance was only small scaled and drift or colonization from unaffected neighbouring reaches is possible. They also show a large variety of life histories with a lot of species being univoltine, contributing to fast recovery rates as reproductive adults or nonsensitive aestivating or resting stages are present more likely after a disturbance event (Niemi et al. 1990).

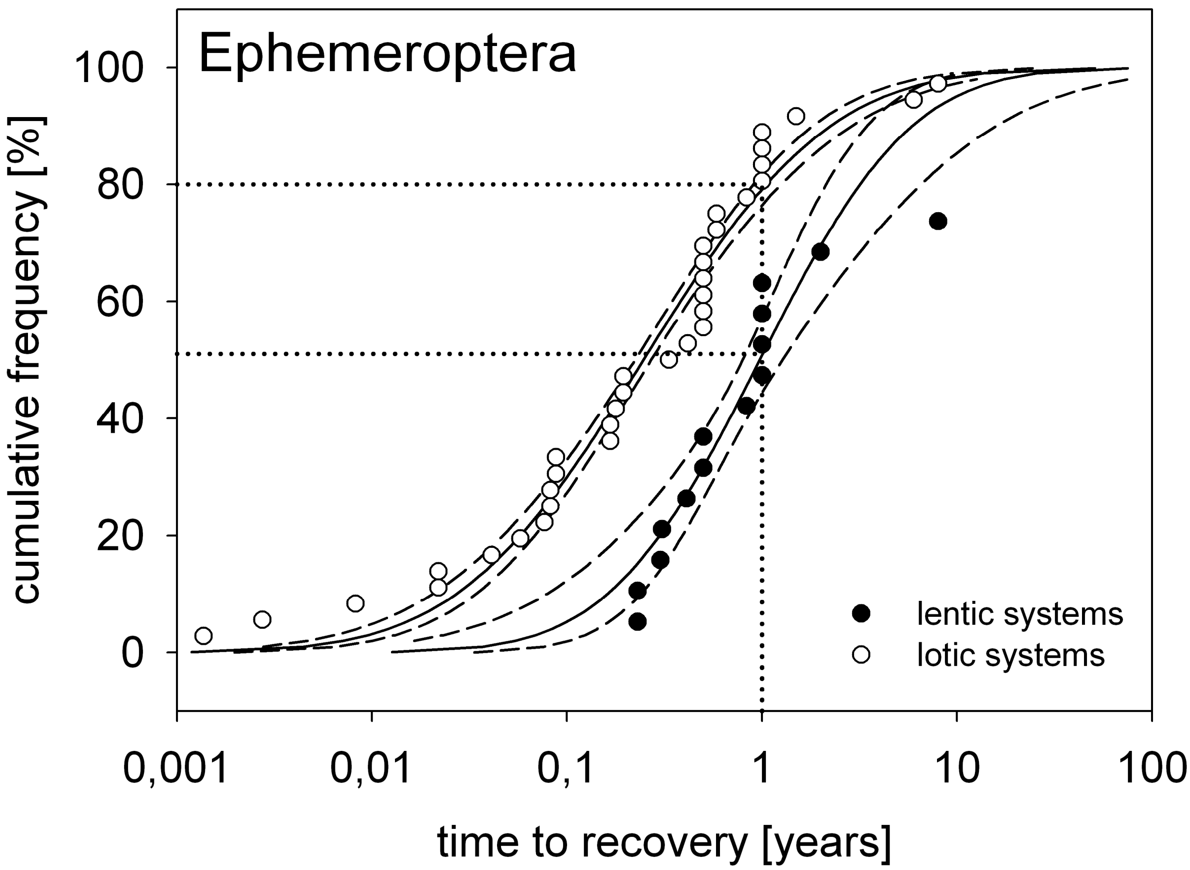

Fig. 8: Cumulative frequency distributions of observed time to recovery of Ephemeroptera after disturbance in lotic and lentic systems. Dots represent data derived from the literature, lines represent regressions of probit analyses in lotic (r^2=0,926) and lentic (r^2=0,802) systems. Legend: see above

Diptera

For Dipterans in lotic ecosystems, a median recovery rate of 0.17 years was calculated from 31 case studies including 54 database entries, with mainly Chironomidae and the lotic family Simuliidae being investigated. In most case studies found, they were the first invertebrates to colonize after a disturbance. For lentic systems a median recovery time of about one year was calculated. Here, 24 case studies with 60 database entries were found.

Fast colonizing, filter feeding, sessile Simuliidae prefer clean stone surfaces with a poor developed organic layer. They are making use of suspended fine detritus that is nearly always present in disturbed streams. Chironomidae are consistently recorded among the first colonizers of new or disturbed subtrates. Like other browsers and gatherers (e.g. mayflies), they are able to exploite the earliest food materials on and among bare substrates (Mackay 1992), they show mainly short generation times with high abundances and gain some relief from poorly colonized habitats after disturbances.

In Flugströmen, a new constructed wetland created for fly fishing in Sweden, Malmqvist (1991) found larval abundances of highly dispersal Simuliidae comparable to the control reach, within 2 month after flooding the new river. Also in Tesuque Creek, recovery of Diptera was completed within 2 month after a flash flood that reduced abundances more than 90% and furthermore, they were the first invertebrates to reach predisturbance densities (Molles 1985). In Mangaotama Stream, chironomids even recolonized within 2 weeks after a flood disturbance (Collier 2003). After 15 weekly applications of permethrin in Sassandra River, Yameogo (1993) observed recovery to pre-treatment state of chironomids within two month, while for Simuliidae it took more than four month.

In cases recolonization was possible via drift (Matthaei 1995, Doeg 1989) or migration of the hyporheic zone (Paltridge 1997) recovery of Diptera was completed within some days up to one month latest. Since the dipteran larvae, especially Chironomidae are highly mobile within the stream bed and prominent in drift, the main factors for colonizing after a disturbance are availability of habitats and food resources (Mackay 1990). If these factors remain unchanged, fast recovery can be expected.

Disturbances that cause long-time effects within the stream may result in longer recovery rates. After the eruption of Mt. St. Helen, that caused a structural degradation within Clearwater Creek, it took two years until chironomids were recorded again (Meyerhoff 1991). Harrel (1985) observed recovery times of seven month for chironomids and nine month for Ceratopogonidae after a cruide oil spill in Turkey Creek. Here, the oil lied undisturbed for at least six days before the cleanup began. This allowed much of the aromatic portion of the oil to evaporate and caused a long-lasting disturbance of the stream-bed with slow recovery of all invertebrates inhabiting the stream. Another example of a long-time affected invertebrate community was recorded by Chadwick (1985). Here, an exposure of mining and milling water for at least 100 years resulted in a total reduction of macroinvertebrates in Silver Bow Creek compared to a reference site. Except for one oligochate no invertebrates were found here. Even though dipterans, primarily chironomids were the earliest colonizers within the recovery process, full recovery did not occur at all during the study duration of eleven years. However, the lack of recovery does not appear to be due to poor habitats, but may be related to presently undetermined amounts of metals still present in the substrate.

For lentic systems several case studies with recovery times of one year or even longer could be found. For example, the colonization of new experimental ponds of the Virginia state University observed by Christman (1992) took one year longest as well as for most investigated dipteran species of new man made Purbeck ponds (Barnes 1983). Here, compared to other invertebrate groups, dipterans colonized rapidly, but the Chironimini were much slower to etablish and some other dipterans even needed two to three years or longer.

When recovery of dipterans after pesticide application was investigated, recovery rates of one year or even longer were observed in several cases. For example, application of fenithothion in Magundy bog ponds, decreased emergence of Chironomidae and Ceratopogonidae by 70-90%, and it took one year until numbers of larvae reached the control level again (Fairchild 1990). Melaas (2001) observed a reduction of Chaoborus of about 80% after the application of rotenone in Minnesota Wetlands. Although Chaoborus is known to be highly reproductive with several generations per year, abundances did not reach pre-treatment level within one year.

Especially for lentic systems, allochthonous colonization seems to be of major importance after a strong effect. If no recolonozation from neighbouring sources is possible, as investigated in artificial mesocosms, full recovery of dipterans is not likely. Caquet et al. (2007) found recovery of Chironominae, Orthocladinae and Corynoneurinae three month after a deltamethrin application latest when colonization was included. If only authochthonous recovery was possible, it took up to seven month for Chironominae and Corynoneuinae to reach the control range while Orthocladiinae did not recover at all. Two other mesocosm studies described by Leeuwanght et al. (1994) and Tidou et al. (1992) showed missing recovery for Chaoborus also since recolonisation was excluded.

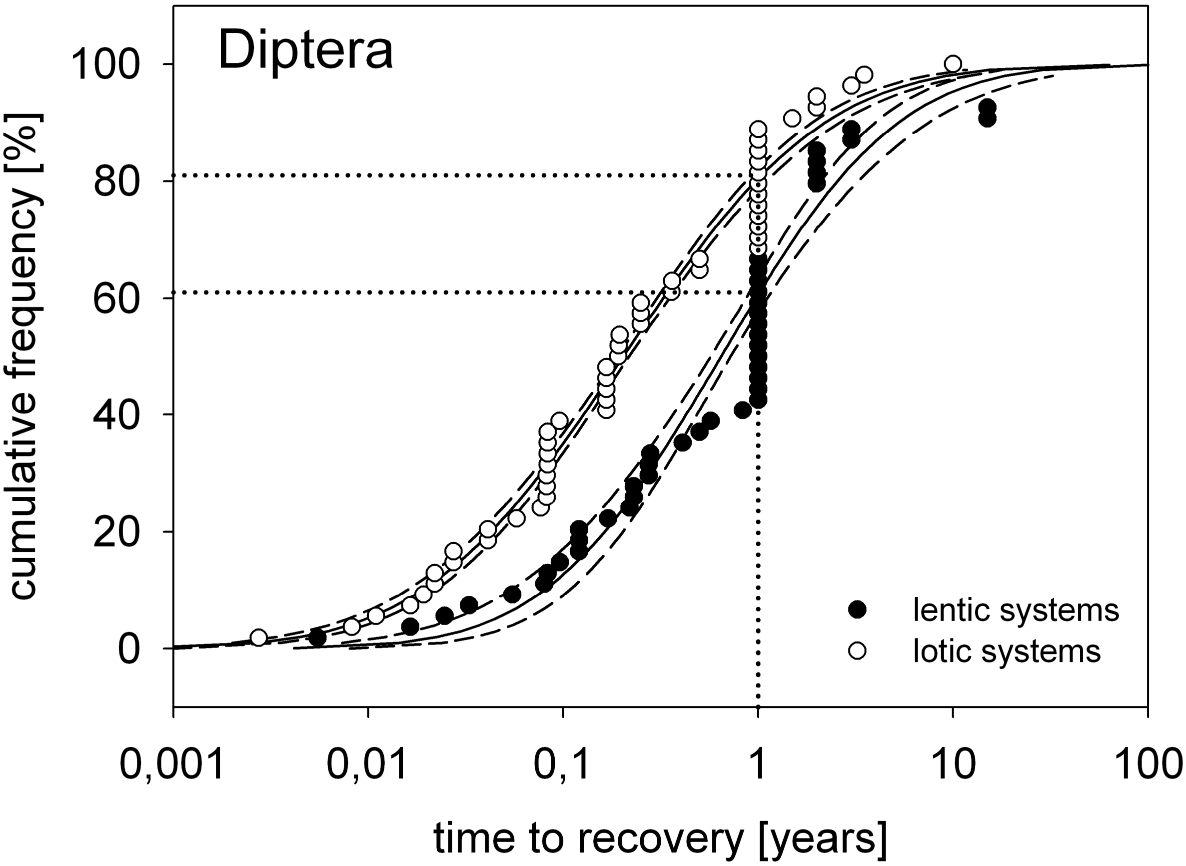

Fig. 9: Cumulative frequency distributions of observed time to recovery of Diptera after disturbance in lotic and lentic systems. Dots represent data derived from the literature, lines represent regressions of probit analyses in lotic (r^2=0,952) and lentic (r^2=0,800) systems (solid: mean, dashed: 95% confidence intervals). Dotted lines indicate the percent of endpoints where recovery was found within one year.

Trichoptera

For Trichoptera we found 20 lotic and 4 lentic case studies including 27 and 11 database entries respectively. Particularly for lentic systems long recovery rates were found resulting in a median time to recovery of two years. Barnes (1983) investigated the colonization of manmade ponds by several caddiesflies and found species dependent differences in colonization times between one and more than 15 years, since adult dispersal from neighbouring ponds was the main pathway for colonization. Two other case studies investigating new constucted lakes showed times for colonizing comparably long. In Veittijärvi lake Trichoptera abundances increased not before 5 years (Danell 1982) and in Kyrkosjarvi reservoirs Trichoptera did not colonize at all within the study period of nine years (Koskenniemi 1994).

Recovery in lotic systems is much faster. For streams a median recovery time of 0.32 years was calculated. However, mainly species belonging to the filter feeding family of Hydropsychidae were investigated (41%), while for example, the major and wide spread family of Limnephilidae was present only with one case study and four database entries. Hydropsychidae are mainly found on rough-surfaced rocks where they establish their nets and retreats. Therefore, they are not always the earliest filterers colonizing, but they have some characteristics of opportunists like a broad diet as they can feed on drifting animals as well as detrital seston (Mackay 1990). Hydropsychidae are part of a typical lake outlet communtiy that was found by Malmqvist (1991) in the new constructed river Flugströmen three month after flooding. Also after heavy metal impacts as recorded by Specht (1984), Smith (2003) and Ryon (1992, 1996), Hydropsychidae recovered within six month latest. In Adair run, larvae of Cheumatopsyche were even the first invertebrates after mayflies to recover after a two years heavy metal discharge that reduced abundances by 50% (Specht 1984). For McCoy Branch, where heavy metal impacts, caused by fly ash discharges lasted over a period of several decades, Hydropsyche already recovered within six month after the coal use had been reduced by 75%. Other Trichopterans followed within the next years (Smith 2003, Ryon 1992, 1996).

After the application of pesticides recovery within a few days up to four month could be observed when drift from upper undisturbed reaches was possible. Yasuno (1982) found full recovery of Trichoptera numbers after application of temephos within less than three month and Diplectrona reached the state of the upstream control after methoxychlor application in the Appalachian Mountain stream within four month after a reduction of 100%. For Pycnopsyche recovery took six month (Whiles 1992). In the North Saskatchewan River recovery of Cheumatopsyche after methoxychlor application was completed even within 5 days (Dosdall 1989).

After a six month drought in Mangela Creek that reduced abundances of caddiesflies by 100% recovery was also completed within 3-7 days. Again colonization was mainly driven by drift from the perennial upper reaches (Paltridge 1997).

For some other cases recovery rates of more than one year were found, mainly after long-time disturbances. In Miramichi River with a yearly application of DDT emergence of some limnephilid caddiesflies could be observed in summer after spraying. Furthermore, glossosomatin cadiiesflies continued to emerge after spraying as they survived as pupae. However, even though a number of insects recovered within 6 weeks up to 2 month, data indicate that real recovery for caddiesflies required up to 4 years or more (Ide 1967). The same trend was found by Watanabe (2000) for Ichi-Kawa River with a heavy metal impact for several hundreds of years. After mine closure, metal concentrations other than zinc were below or near the detection limit and Hydropsychidae started to recover fast. However, taxa combination reached a stable level not before 5 years (Watanabe 2000). In another case the eruption of Mt. St. Helen caused a physical disturbance and an aprupt change in the character of the Clearwater Creek. As the structure showed slow recovery, Trichoptera did not reestablish before 3 years (Meyerhoff 1991).

Since most case studies found investigated fast recovering, filter feeding Trichopterans little can be said about other feeding types as shredders and scrapters that are also common within this taxonomic group. Since both, shredders and scrapers have seasonally bound life cycles depending on the availability of food, recovery within this taxa might be limited when disturbance is unexpected (Mackay 1990).

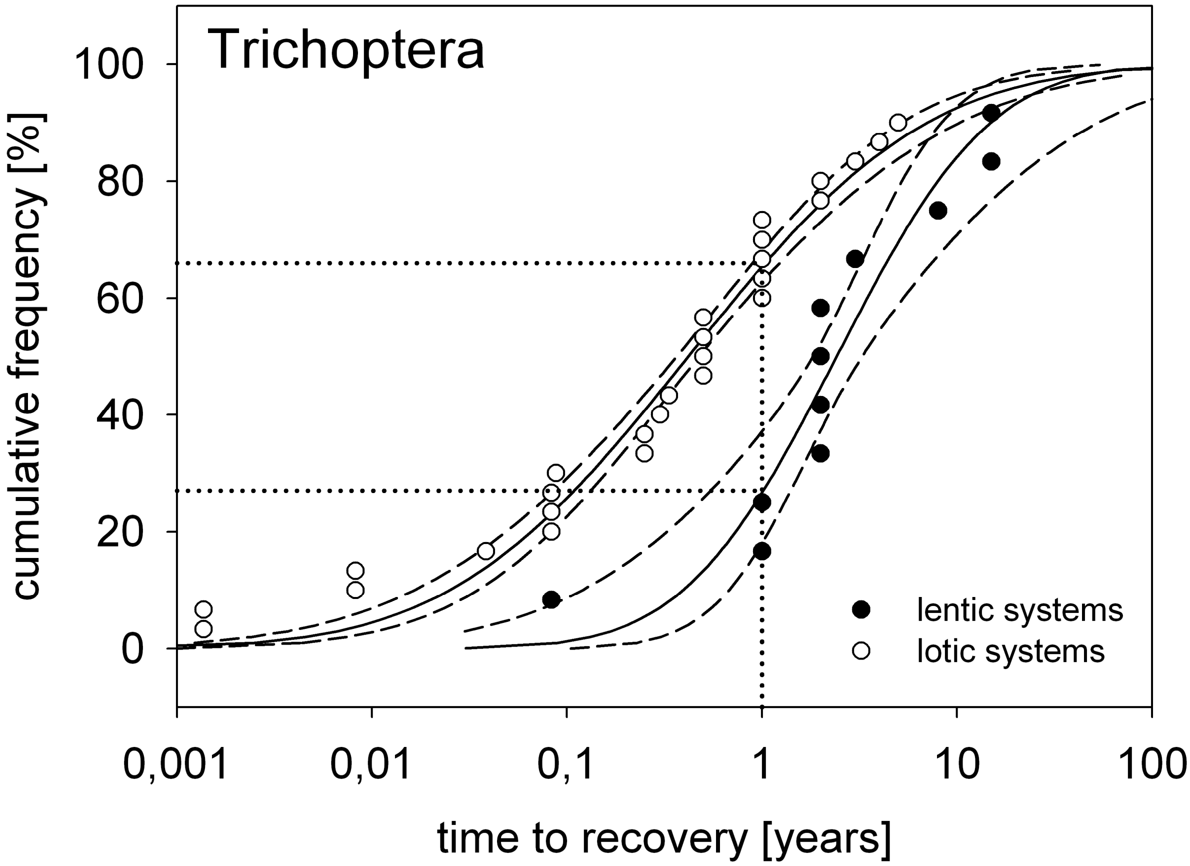

Fig. 10: Cumulative frequency distributions of observed time to recovery of Trichoptera after disturbance in lotic and lentic systems. Dots represent data derived from the literature, lines represent regressions of probit analyses in lotic (r^2=0,926) and lentic (r^2=0,785) systems. Legend: see above

Plecoptera

Plecoptera were only investigated in 11 stream cases and 1 artifical stream experiment. In summary 13 database entries were found with a median recovery time for lotic systems of 0.29 years.

Very fast recovery within one month latest was found within three case studies (Brooks 1991, Harriman 1988, Morrison 1990, Matthaei 1994). For Finniss River Brooks (1991) found stoneflies being fast colonizers that attained predisturbance densities even after 1 day. Since this case study investigated the effect of substratum particle size on recolonization by benthic macroinvertebrates with small plot experiments, recolonization appeared to be by surface movement (vertical migration) from neighbouring still intact areas. The experiments showed that the results from small-scaled disturbances, as inverstigated here, cannot be extrapolated to a larger-scaled scenario, because the scale of disturbance strongly influences the rate and pathway of recolonization as well as the source and faunal composition of the recolonists. Another case study with recolonisation times for Plecoptera from7 to 30 days was described by Matthaei (1994). In the river Necker physical disturbance by a natural flood as well as experimental disturbance of the streambed reduced stonefly numbers by 90%. In both cases, invertebrate drift from upper undisturbed reaches was probably the most important pathway for recolonization. For Loch Erd Forest, an acid stream with low pH values from about 4.5 to 6.7

highest and a naturally low species richness recovery of stoneflies was found one month after a drought that caused an effect of 100%. The fast reappearence of Plecoptera after the stream refilling was suggested to be driven by animals that survived the drought as eggs or by oviposition soon after the water level rose (Harriman 1998, Morrison 1990).
Disturbances on a larger scale or without nearby refuges for recolonization showed longer recovery times. For a the new constructed Flugstömen stream Malmqvist (1991) found some Plecoptera colonozing after 5-6 month, whereas other species were not recorded after 2 years due to their poor dispersal ability. For Glacier Bay, another constructed wetland fed by meltwater, it even took 10 years until Plecoptera reached the state of the control (Milner 1987, 1999, 2000, Flory 1999 a,b).
After the physical disturbance of Tesuque Creek by flash flood that reduced total number of invertebrates more than 90%, recovery of stoneflies took 9-12 month (Molles 1985).
Within two case studies fly ash discharges for a longer time period resulted in a reduction of invertebrates. In McCoy Branch stoneflies were completely absent after a discharge of about 30years and reached the state of the controls not before three years (Smith 2003, Ryon 1992, 1996), while Specht (1984) did not observe full recovery of Adair Run within the study duration of 2 years. Here, fly ash discharge lasted about two years with a reduction in stoneflies of about 50%.
In two other cases, the application of pesticides was investigated, both resulting in an effect of almost 100%. Here, recovery of Plecoptera could not be oberserved within the given study duration of 3 and 7 month respectively (Yasuno 1982, Beketow 2008).

Fish

The literature search found 20 case studies with 44 data base entries for the recovery of fish species or fish communities in lotic systems, most of the case studies were located in north america For lentic systems only one case study was found. The median recovery time for fish were 0.3 years.

In Europe, most fish species are expected to have one reproduction cycle per year. Only under Mediterranean conditions some small species might have more than one generation per year. Thus, it is obvious that short-term recovery of fish population after pesticide stress can not be expected by population growth. However, several examples have shown that emptied stream sections are quickly re-occupated by individuals searching for territories.

No recovery was observed in 4 case studies. In all these studies fish were effected by 100% and the stressors included physical disturbance (Gunning 1969, Hawkins 1990, Lonzarich 1998) and pesticides (Olmsted 1974). In one studie (Olmsted 1974), investigating the effect of an unknown pesticide, all fish species recovered except one. The rapidly repopulating species were eurecious and vagile (e.g. *Notropis boops*), while the more slowly repopulating species (*Etheostoma zonale*), which do not recovered within 1 year, were stenoecious and less vagile (Olmsted 1974). In another study (Lonzarich 1998), that only lasted 40 days, no recovery to the pre-treatment abundance was observed in isolated pools for some species, whereas in connected pools recovery of all fish could be observed within the study period. One study (Gunning 1969) compared the recovery rates of a single species (*Erimyzon tenuis*) in two streams and found recovery within 13 month in one stream but not in the other. The observed differences in recovery pattern could not be explained by the authors. Another study (Hawkins 1990), investigating the recolonisation of streams after the eruption of Mount St. Helen, found recovery for sculpins within 5 years, for which refugia exist upstream, but not for trouts within 10 years, because no refugia were present and recovery from downstream was blocked by a waterfall, additionally the substrate changed and suitable spawning areas for trouts diminished.

Slow recovery, above 5 years, of fish was observed in 5 case studies, including the river rhine after the disturbance with pesticides and other contaminants after the burning of “Schweizerhalle”. In the river rhine 40 of 47 fish species recovered after 4 years (Lelek 1990), after the high contamination which excerpts dramatic effects on the whole biocenoses up to 400 km downstream. The not recovering 7 species showed reduced abundances already before the accident (Lelek 1990). Within two other studies (Hanson 1974, Hawkins 1990) the effects observed on fish species, due to flood, eruption of Mount St. Helen, were accompanied by changes in habitats and reduced immigration possibilities. In some studies knockdown of a specific species led to invasion of new species, not present before (Hanson 1974). The remaining two studies (Diamond 1993, Ryon 1996) investigated the recovery after long lasting heavy metals disturbances. For all other studies average or fast recovery of fish species below 1 year, mostly below 0.5 years were found.

These findings are in line with a previously published review (Detenbeck et al. 1992) that was based on a broader database of recovery processes and dynamics for fish. Within this review the authors concluded that lotic fish communities are not resilient to press disturbances (e.g., mining, logging, channelization) in the absence of mitigation efforts (recovery time >5 to >52 yr). Here, recovery was limited by habitat quality. Following pulse disturbances, autecological factors, site-specific factors, and disturbance-specific factors all affected rates of recovery. Centrarchids and minnows were most resilient to disturbance, while salmonid populations were least resilient of all families considered. Species within rock-substrate/nest-spawning guilds required significantly longer time periods to either recolonize or reestablish predisturbance population densities than did species within other reproductive guilds. Recovery was enhanced by the presence of refugia but was delayed by barriers to migration, especially when source populations for recolonization were relatively distant.

The studies were mainly conducted in North America, and thus there is some uncertainty if these findings can be transferred to Europe. Nevertheless recovery of fish species were found to be mostly due to immigration, and where recolonisation was hindered recovery was significantly prolonged. Therefore for lentic systems it is expected that recovery of fish is remarkedly lower, since immigration is less probable.

Discussion

Previously published reviews (Niemi et al. 1990, Yount & Niemi 1990, Wallace 1990) concluded that factors affecting the recovery rate of the biota in aquatic systems can be considered as independent of or dependent on the taxonomic groups found within the system. Generation time, fecundity, presence of aestivating or resistant stages, propensity to dispers and predation competition interactions were identified as important dependent factors (Niemi et al. 1990). Factors independent of the organisms that affected recovery rates were hydrological factors or general changes in habitat, changes in system productivity, residual toxicity, time of impact and presence or distance of refugia (Niemi et al.1990, Yount & Niemi 1990, Whiles & Wallace 1995). Correspondingly, in the following the results gathered from the literature search will be discussed by taxonomic factors first and by factors independent of organisms in a second step.

With the exception of algae, where only few other information was available, total density and biomass of higher taxonomic groups (e.g. macroinvertebrate) was excluded from the analysis of the current review, as Niemi et al. (1990) concluded that time to recovery of these composite parameter and these of individual taxa might differ. Furthermore, functional recovery and taxonomic recovery will not necessarily be given within the same time frames (Wallace 1990) and will be discussed separately along taxonomic examples.

Summary of patterns and times to recovery found for taxonomic groups

In the present review, recovery times after disturbance vary from < 1 month to > 16 years, depending on several factors like taxonomic level, life history traits and sensitivity to the stressor.

Algae and zooplankton were able to recover within one year in nearly all case studies identified. Due to their short life cycle and high reproductive potential it can be concluded that there is only a minor risk of extinction due to short pesticide applications for these groups. Nevertheless, total extinction even of these groups may not result in fast recovery, if resting stages or refuges are not present for the specific ecosystem. Algae were mostly evaluated on community level or functional group only, therefore no conclusion on species level is possible. However, the phytoplankton community is a very dynamic system. Populations might appear in considerable abundance only for short time before replaced by other species. Thus, recovery of single species within the time frame of typical studies (e.g. mesocosms) can not be monitored. The dynamics of the phytoplankton can also lead to shifts in the community structure as a result from direct or indirect effects of a stressor. It is known that pollution can change algae community structure, selecting non sensitive species, without changes in biomass or chlorophyll content probably known as pollution induced community tolerance .

Within the group of lotic macroinvertebrates, Niemi et al. (1990) found at the ordinal level the following ranking of time to recovery for the major groups in lotic systems: Diptera < Ephemeroptera < Trichoptera < Plecoptera. This holds true also for our analysis taking the median recovery time into account (Fig. 4), where we found the same detailed pattern of recovery previously described by Mackay (1992). Chironomidae are consistently recorded among the first colonisers (Yasuno 1982, Chadwick 1985, Milner 1994, Pires et al 2000, Churchel & Batzer 2006). Among the mayflies, Baetidae (mostly *Baetis sp*.) were highly abundant, early colonizers (Lake & Doeg 1985, Arnekleiv & Storset 1995, Smith 2003) and certain Leptophlebiidae (e.g. *Deleatidium sp*.) were similarly recorded (Sagar 1983, Scrimgeour 1988). The early arrival of the browsing and gathering dipterans and ephemeropterans mentioned above is consistent with their ability to exploit the earliest food materials on and among bare substrates. Later in the recovery of streams grazers and shredderers contribute to colonization. If colonists cannot browse or gather filter feeders are the alternative early feeding types. Accordingly, Simuliidae, Hydropsychidae and some chironomine chironomids (e.g. *Rheotanytarsus sp.*) were found to re-appear quickly in disturbed patches (Chadwick 1985, Malmqvist et al. 1991, Yaméogo et al. 1993, Matthaei et al. 1996, Weng et al. 2001), whereas shredder-dominated communties, mostly typical in first-order forest streams, are rather slow to recover after disturbance (Mackay 1992). Indeed, shredderers have often been reported as later colonizers than other detritivores and herbivores (summarized in Mackay 1992), since they often have poor dispersal abilities. With the exception of *Gammarus*, shredders are rather rare in drift, so they are not routinely transported to disturbed reaches from upstream sources. Shredder life cycle tend to be long and tied to seasonal inputs and decay of CPOM. Many life cycles are synchronous. Therefore, colonizing stages may not be available following an unexpected disturbance.

Concluding from the presented data, lotic invertebrate communities that are adopted to frequently disturbance most likely include substantial portions of baetid, leptobhlebiid, and sometimes heptageniid mayflies, multivoltine black flies, opportunistic chironomids, and Hydropsychine caddiesflies. Depending on the recovery rate of periphytic algae grazers will colonize soon after detritivores. These findings are in line with Mackay (1992). Coleoptera were not represented by a large number of endpoints in previous reviews (e.g. Niemi et al. 1990). However, in our analysis we found times to recovery similar to these of Trichoptera and Plecoptera.

Within bentic invertebrates Ephemeroptera and Diptera show high drift rates resulting in high recovery potentials (e.g., Townsend and Hildrew 1976). In addition, dipterans and ephemeropterans have a large variety of life histories (i.e., univoltine, muhivoltine, synchronous emergence, asynchronous emergence), but it has to be stated that most ephemeropteran data was available for baetids. Due to their short life cycle Baetidae are often among the the first species recovering (Mackay 1992), compared to other lotic ephemeropterans, e.g. *Ephemera danica* exhibiting a 2 year life cycle. For example Methoxychlor input in a small headwater stream of the Appalachian mountains led to the reduction of aquatic insect abundances and biomass by 90% (Wallace 1989). Functional recovery occurred within 2 years while taxonomic recovery required about five years (Whiles 1995). Several ephemeropteran and plecopteran taxa were among the least resilient taxa and did not recover until three to five years following the treatment. Thus, differences in recovery can be large within these groups. And Baetidae might not be sensitive for the whole group.

Interestingly, recovery of endpoints describing biodiversity like taxa richness, community composition and diversity indices last longer than endpoints for single species (Fig. 11). This astonishing fact can be explained e.g. by methodological reasons. Recovery is mostly measured for high abundant or selected species (e.g. Baetidae), whereas in the analysis of taxa richness, community composition or diversity indices also the low abundant species are taken into account. The later also consider the equal distribution of species, and therefore the recovery of the low abundant groups. This example shows that overall recovery of ecosystems might be longer lasting than can be expected from literature studies observing single species abundances.

For some macroinvertebrate groups or taxa missing a high dispersal potential (e.g. Oligochaeta, Molluscs and some Coleoptera), higher times to recovery and no differences in recovery times between lotic and lentic systems were found. This fact will be discussed in the following section.

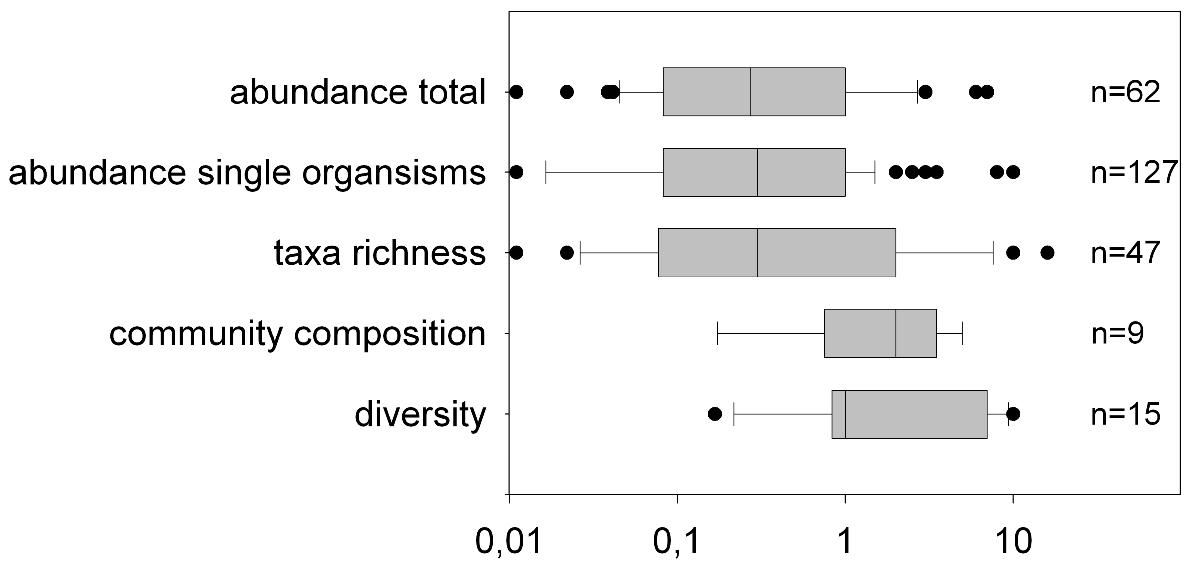

Fig. 11: Time to recovery of community measures in lotic macroinvertebrates. Taxa richness includes recovery in overall macroinvertebrates or selected taxonomic subgroups, community composition includes principal response curves and indices of similarity; diversity integrates different diversity indices

Summary of factors driving recovery independent of taxonomy

Previous reviews (Niemi et al. 1990, Wallace 1990) concluded that a number of taxonomy independent factors contribute to the rate of recovery of a population, namely the spatial scale of disturbance, the persistence of the stressor, the riming of contamination in relation to the life history stage and habitat and landscape related aspects like the presence of refugia within the disturbed system, connectivity to and the distance of sources for recolonisation.

In general, the taxonomic dependent factors regulated recovery time within a relatively narrow range, but long term effects were observed whenever a species was fully eliminated and if barriers to recolonization were present (e.g. Hawkins & Sedell 1990). Furthermore, populations will develop and stabilize sooner if the disturbance is a local rather than a regionally extensive catastrophe. Recovery in lotic systems is faster if undisturbed sites are present upstream or refugia present in the affected reaches (Cuffney et al. 1984). Drift is one of the dominating recolonization and recovery pathways within lotic systems (Brittain & Eikeland 1988); recovery due to downstream sites is much slower since it needs active movement of the organism. This implies that fast recovery can only be achieved if the species of interest are present in upstream reaches or refuges. Only for fish the active movement is more dominant than drift (Detenbeck et al. 1992). Additionally, disturbance that resulted in physical habitat alterations were the most common impact associated with long recovery times (Niemi et al. 1990). This pattern was also found in the current analysis, in which recovery within lotic systems was significant lower than in lentic systems, which are not connected to unimpacted sites.

Stressors for aquatic ecosystems are grouped as pulse or press disturbance. Whereas recovery after pulse exposure might be fast, recovery after press disturbance can last for years as shown e.g. in the case of fish recovery following the eruption of St Mount Helens (Hawkins & Sedell 1990). As defined by Yount et al. (1990) spills of nonpersistent chemicals (e.g. pesticides currently in use) typify a pulse disturbance, whereas long-term pollution or clear-cutting of a forested watershed typifies press disturbances to lotic ecosystems. Recovery from the effects of a pulse disturbance appears to be rapid, with the exception of taxa exhibiting long life cycles and low dispersal, as long as the stressor is localized and recolonisation sources are accessible. For press disturbance, which results per definition in alterations of the habitat (Yount et al. 1990),, long-term impacts on lotic systems were frequently found.

In the current review two distinct groups of stressors could be extracted from time to recovery data. Within the first group, represented by four types of stressors, physical disturbance, flood, drought and pesticides, recovery within one year was observed in around 80% of the macroinvertebrate database entries. In the second group, organics, metalls and constructed wetlands are clustered. For this group more than 50% of the macroinvertebrates did not recover within one year.

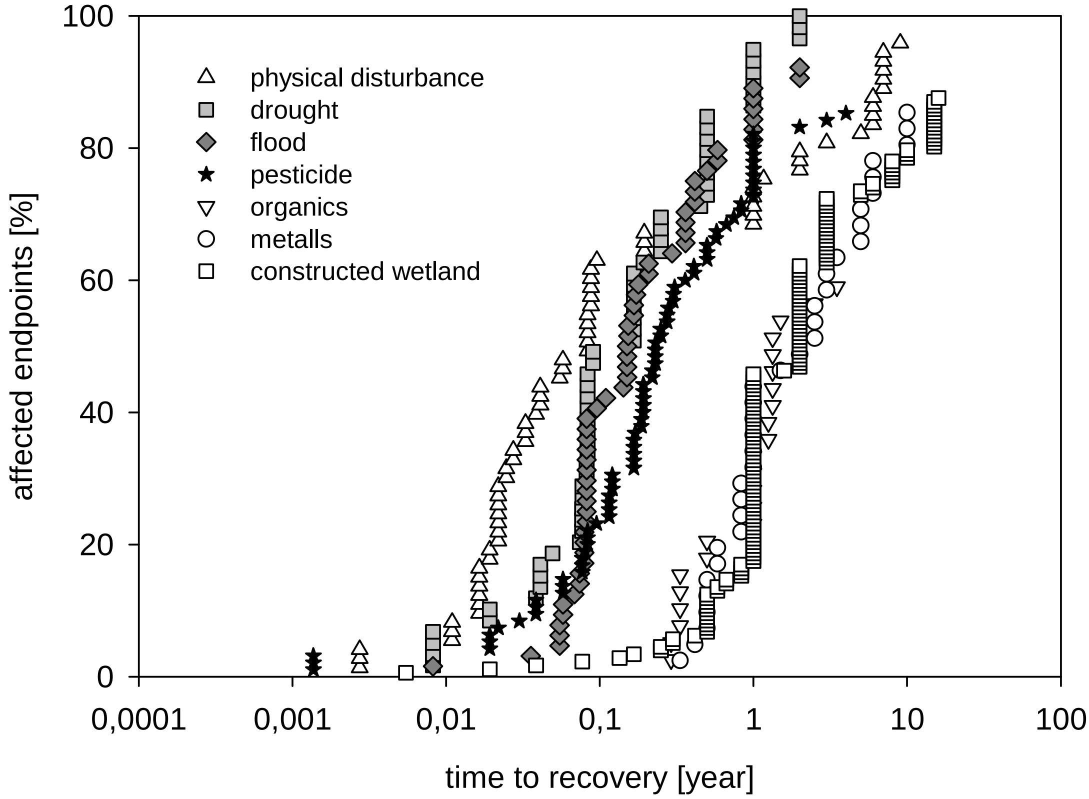

Fig. 12: Cumulative affected endpoints for all macroinvertebrates as a function of time to recovery after stress in lotic and lentic systems. Dots represent data derived from the literature, grouped by type of stressor.

In many field studies and during registration of pesticides pulse disturbances of single pesticides were applied in order to assess effects and times to recovery. These studies assume that pesticide exposure is a periodic short term event, and will therefore result in short term effects.

In the field, intensive use of e.g. insecticides, fungicides and herbicides might result in press disturbances and long term community shifts as shown for the fruit orchard region Altes Land near Hamburg, Germany (Heckman 1982, Schäfers et al. 2006) and streams in the region of Braunschweig, Germany (Liess & von der Ohe 2005). Heckmann (1981) found long-term effects of the chronic use of different pesticides fungicides, insecticides, acaricedes, herbicides from the 1950th through the 1970th. Many aquatic species became completely resistant to the inputs of agricultural chemicals, while others were eliminated from the habitat over a period of 25 years. This example and other case studies (e.g. Wallace 1989, Whiles 1995) implies that pesticide use, which is in a single event a pulse disturbance, can have due to multiple applications and mixture toxicity become a press disturbance in aquatic ecosystems.

Floods and droughts might also appear as both pulse and press disturbances depending on their frequency of occurrence. However, these kinds of disturbances are often attributed as regular endogenic features (Reice et al. 1990). The inherent ability of a lotic ecosystem to recover is determined by physical characteristics of the system and life history characteristics of the organisms in the system (Yount & Niemi 1990). Thus, another confounding factor for the extrapolation of the recovery patterns found is the fact that communities might be apparently selected for life history traits (e.g., rapid development, continuous emergence, and diapausing eggs) that facilitate rapid recolonisation (Gray 1981, Fisher et al. 1982) in disturbed systems, e.g. for regularly flood and drought or pesticide use (Sousa 1984, Wallace 1990, Yount & Niemi 1990). This can lead to faster recovery rates for the investigated, probably adapted communities and might be not protective for undisturbed communities. However, it is not expected that undisturbed communities will be found in the current agricultural landscape. Therefore a detailed definition of the protection goal for pesticide risk assessment is needed (Brock et al. 2006).

Relevance for a geo-referenced probabilistic risk assessment of pesticides

A considerable number of investigated studies were not intended to yield time to recovery informations. This results in a heterogenous dataset in which the selected endpoints (study duration, taxonomic classification level or reference), or the description of the stressor and landscape were not suitable for detailed analysis. Interestingly most studies focus on the same species within a group, e.g. *Baetis sp.* as Ephemeroptera or *Hydropsyche* sp. as Trichoptera. Data for other species are mostly not reported and it remains unclear if other species were not investigated, not evaluated or do not show effects or recovery.

Although, by comparing colonizers of experimental patches, disturbed reaches, newly watered channels and disturbed rivers around the world Mackay (1992) found identical patterns of recovery on family or generic level, extrapolating results from North America to Europe, with different species and climate conditions, remains questionable. Furthermore, recovery due to recolonisation cannot be studied in most artificial systems, like mesocosm or stream mesocosm, since recolonisation is mostly hindered due to the study design. Thus, for a more detailed consideration of recolonisation in environmental risk assessment, more and consistent longterm field data on the dispersal and recolonisation potential of single taxonomic groups are needed.

Confounding factors for the use of recolonization potential gathered from literature studies for pesticide risk assessment are the following. Usually the recovery is analyzed in the absence of stressors (e.g. after dissipation of the test item in a mesocosm study or a field test). This is adequate for the usual risk assessment related to the registration of one specific active substance or product. However, in the agricultural landscape recovery after exposure to one pesticide can be affected by other stressors, e.g. exposure to other pesticides. Most data were found for new constructed wetlands, but the analysis showed that this type of stressor is not comparable with recovery of pesticides. Therefore most data gathered from literature can not be used to predict recovery or recolonization after pesticide disturbance.

For the prediction of recovery within the risk assessment of pesticide three main factor groups have to be taken into account: 1) stressor related factors 2) species related factors 3) spatial factors. The stressor related factors, like dissipation time as well as intrinsic toxicity measured in the standard ecotoxicological tests are the basis of current risk assessment schemes. the ecology of different species is considerd less frequently, e.g. by the use of mesocosm studies. Trait based approaches might improve our ability to extrapolate effects and recovery patterns between species in different (model) ecosystems (e.g. Baird et al. 2008).

The factor remaining is the effect of the habitat and landscape structure which drives the spatial distribution of exposure and thus of effects as well as the presence of sources for recolonisation. This consideration can only be made in a GIS-based manner, if all neccesary data is available. First of all the data from this review clearly indicates that lotic and lentic systems differences in their recovery potential and therefore have to be separated within risk assessment. Additionally it was demonstrated in the current and previous studies (Yount et al. 1990, Niemi et al. 1990, Wallace 1990) that immigration, e.g. via drift, is a recovery driving factor in particular for species with long life-cycles. This was also pointed out by Barnthouse (2004) in a theoretical study, were time to recovery for mysids were reduced by 50% if 1% immigration per day was assumed. Potential of drift and movement of various species can be estimated from available literature data (Englund et al. 2004). Whereas several approaches already exist to model recovery depending on factors related to the stressor and species (for an overview see Galic et al, submitted), less examples are available where habitat diversity (refuges) and/or landscape structure (presence of sources for recolonisation) have been considered (e.g. Sherratt & Jepson 1993, Spromberg et al. 1998, van den Brink et al., 2007). Whenever recolonisation is considered in pesticide risk assessment, these obstacles have to be solved.

Therefore it is necessary to conduct special field monitoring programs to investigate the recovery and recolonisation potential of single species and communities within European streams, ditches and ponds related to habitat diversity and connectivity of water bodies in the landscape. To achieve a consistent dataset from these monitoring studies a guidline should be developed. This is the main conclusion from all reviews about recovery since 1977 (Cairns et al. 1977 (in: Yount et al. 1990), Fisher 1983, Niemi et al. 1990, Wallace 1990, Bond 2008), indicating no significant progress within this topic over the last three decades.

These studies should be last until recovery is reached or at least 1.5 years, because otherwise they are not usable for the given question. Sampling should be conducted at least monthly to catch the recovery for all taxonomic groups adequately. Endpoints should be recovered for each species compared to a reference or predisturbance state. Valuable examples for this questions were also found in the literature search (Liess & Schulz 1999).

It is not expected that this data gap is closed by other monitoring studies, like the Water Framework Directive (WFD), since other questions led to other study designs not suitable for the question. For the WFD general information is gathered about a huge amount of sampling sites usually in streams of higher order. For the question of recovery of aquatic populations after pesticide stress in edge-of-field-water bodies, less sites would have to be sampled more frequently.

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