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Mitteilungen aus der Biologischen Bundesanstalt
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Workshop on Risk Assessment and Risk Mitigation Measures in the Context of the Authorization of Plant Protection Products (WORMM) 27.-29. September 1999

Organized by the Federal Biological Research Centre for Agriculture and Forestry, Biology Division, Braunschweig, Germany

Edited by

Rolf Forster
Martin Streloke

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Opening remarks

Klingauf, F.

President of Federal Biological Research Centre for Agriculture and Forestry (BBA), Messeweg 11/12, 38104 Braunschweig, Germany

Background and objectives

In January 1999 the BBA together with the authorities of the NL decided to organize a workshop in order to bring together experts from neighboring EU-member states, experienced in the different scientific disciplines relevant for decision-making especially in the field of risk assessment and risk mitigation measures, for an exchange of views.

The main objectives of the workshop should be

- to demonstrate risk assessment and risk mitigation schemes currently applied in the context of the authorization of plant protection products in different countries (aquatic and terrestrial case studies),
- to describe the contamination of non target areas and effects on non target organisms by plant protection products under realistic field situations,
- to introduce new approaches on how to overcome shortcomings of currently used strategies especially in the field of risk mitigation measures.

Therefore we titled our workshop “BBA-Workshop on Risk Assessment and Risk Mitigation Measures in the Context of the Authorization of Plant Protection Products” or to make a long title short “WORMM”.

Braunschweig, September 1999

Prof. Dr. Fred Klingauf

New approaches in risk assessment for pestcides

Probabilistic risk assessment of agrochemicals

Solomon, K.R.

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Abstract

Concern for the environment has resulted in greater scrutiny of both old and new plant protection products and increased efforts have been directed to developing more rigorous but more realistic procedures for the ecotoxicological risk characterization of these agrochemicals. These techniques include probabilistic analysis of toxicity and exposure data. The ecological basis for these risk assessments has been broadened though a better understanding of the relationship between structure and function in populations of wildlife and the role of keystone species in maintaining ecosystem functioning. This understanding has been incorporated into risk assessment through a better understanding of population recovery rates and functional resiliency.

The use of probabilistic approaches has improved our ability to combine toxicity data for many species into ecological risk assessments. Distributional analysis of toxicity data has allowed the better characterization of toxicity data for pesticides. Distributions of exposure data can be combined with distributions of toxicity data to assess ecological risks. These probabilistic risk assessment methods are being assessed for incorporation into risk assessment procedures in a number of regulatory jurisdictions.

Introduction

Pesticides are specifically used for the control of organisms for the protection of crops, human health or structures. Use in these circumstances always occurs after some form of risk assessment has occurred in relation to the particular situation. For example, an agriculturalist may consider the cost of the pesticide used to control an infestation of fungi in a fruit crop against the benefit resulting from increased value of the crop to the ultimate consumer, be this in improved quality or better storage properties. These types of decisions have been made for thousands of years in agriculture. Integrated pest management (IPM), as commonly practiced in agriculture involves similar risk management decisions, although the risks are not as easily measured and the benefits are more indirect. Risk benefit decisions made during pest control or pest management operations are internalized to the act of agriculture or crop production (Figure 1). These decisions are made with very specific uses in mind. The history of the crop or animal production system is known and the particular pest situation may be known in great detail, especially in cases where pest numbers and life table information is available, such as in intensive IPM practices where scouting of pest populations and descriptive information on climate and other environmental factors are collected. Decisions made in agricultural production do not specifically consider the effects of the pesticide outside the area of specific use, such as may result from movement away from the area of application to areas that support non-target organisms or to non-target organisms that utilize the agroecosystem as habitat (birds, mammals, or other terrestrial organisms). Assessing the importance of these external risks is the focus of this paper.

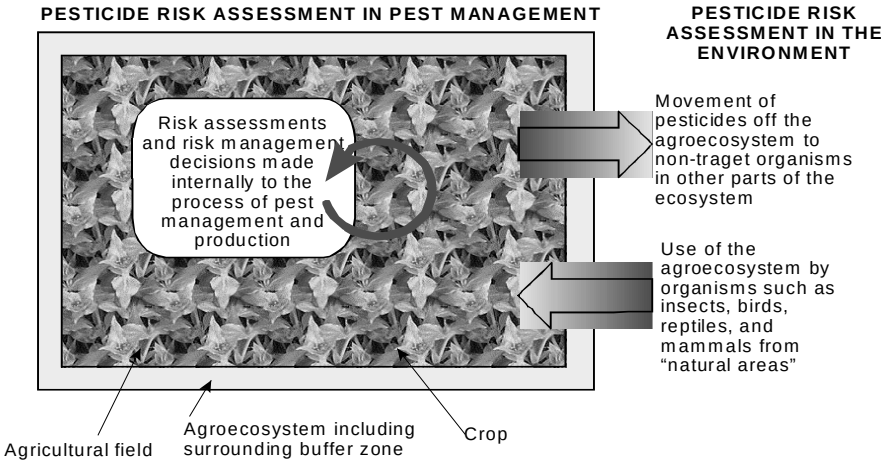

Figure 1 Illustration of risk assessment and risk management decisions in pesticide use in agriculture

Structure and function of ecosystems in relation to risk assessment of pesticides

Assessment endpoints and measures of effect can be defined at all levels or organization in ecosystems, from that of the individual to that of the community. However, these are not necessarily of equal importance (SUTER et al., 1993). In contrast to human health protection, individual organisms in the ecosystem are generally regarded as transitory and, because they are usually part of a food chain, are individually expendable (SUTER et al., 1993). A self-maintaining or reproducing population is persistent on a human time scale and can be easily conceptualized by humans as being in need of protection. Thus, most assessment measures in ecological risk assessment are defined at the population, rather than at the organisms level. Only in the case of the protection of rare, endangered, or long-lived species are organisms in the environment afforded similar protection to that enjoyed by humans. Generally, ecological risk assessment is aimed at protection of the functions of populations, communities and ecosystems. This acknowledges the fact that populations are less sensitive than their most sensitive member and, likewise, that communities and ecosystems are less sensitive than their most sensitive components. Effects on a population are not necessarily of concern (to the ecosystem) as long as the functions the population can be taken over by other organisms. In this context, function is the interaction of the population with other populations or the abiotic environment. Functions in ecosystems are normally related to energy and nutrient flow: production of biomass, consumption of biomass, controlling the abundance of other species, providing food to predators, or processing organic detritus, and mineralizing organic compounds (SUTER et al., 1993).

Functional redundancy is essential to the continuance of ecosystems in the face of natural stressors, such as, for example, the effects of winter in temperate climates. Redundancy is the result of selection imposed by fluctuating and unpredictable environmental conditions. Most ecosystems exhibit functional redundancy, where multiple species are able to perform each critical function (BASKIN, 1994, WALKER, 1992, 1995). Functional redundancy is particularly relevant to the ecotoxicological risk assessment. It is the basis for being able to tolerate effects in some sensitive populations as these are unlikely to impair the functions of the ecosystem as a whole. This is the basis for being able to tolerate some species being affected, such as in setting water quality guidelines (STEPHAN et al., 1985). As illustrated in Figure 2, there is a general relationship between exposure concentration and impact of any substance, however, there are deviations from this general rule. For example, functions may be maintained where few species are affected but, as the number of species affected increases, indirect effects amplify the effects of the substance to greater than predicted levels. Redundancy of function has been observed in a number of experimentally manipulated systems ranging from terrestrial (TILMAN, 1996; TILMAN et al., 1996) to aquatic (GIDDINGS et al., 1996;

SOLOMON, 1996; STEPHENSON et al., 1986). These observations support the concept that, in ecotoxicological risk assessment, some effects at the level of the organism and population can be allowed, provided that these effects are restricted on the spatial and temporal scale. In other words, they do not affect all communities all of the time and that keystone organisms are not adversely affected. In the context of selecting assessment measures, it has become increasingly recognized that these should be at the functional level of populations and the community and that some effects on populations and species diversity may thus be tolerated.

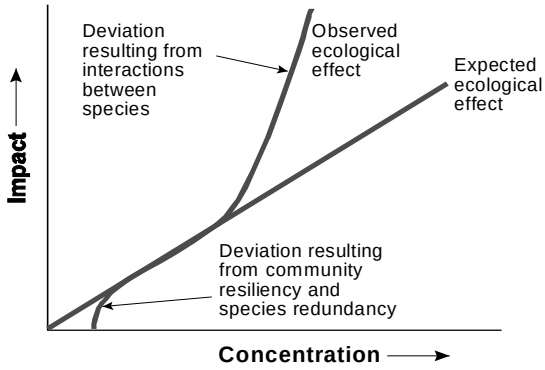

Figure 2 Illustration of ecosystem resiliency in response to stressores and effects caused by interactions between organisms

The risk assessment analysis

Two important steps in risk assessment of pesticides are characterization of the effects of the pesticide and estimating exposures to the pesticide.

Characterizing effects

Standardized test methods for pesticide risk assessment are routinely used and required by a number of regulatory agencies. The basic principle behind the use of standardized laboratory toxicity tests is not that the particular organisms in the test are those that require protection in the environment but rather that these organisms act as surrogates for all those other organisms in the ecosystem that could be exposed but which, for one reason or another, cannot be tested in the laboratory. Because of this, test organisms are usually selected for ease of use and because historical testing has shown that the species is particularly sensitive and would therefore provide a worst case measure of effect. To make the effect measure even more conservative, the tests are normally conducted under conditions where the exposures are maintained at a constant concentration, usually by continuous addition to a continuous flow treatment system.

Population-level assessment procedures (those carried out in populations of single species such as in chronic laboratory tests in organisms with a short life cycle) cannot take into account effects that involve interactions between populations of different species in communities or those that affect ecosystem function, such as recovery and changes in productivity or nutrient flow. A number of procedures have been proposed for ecosystem and community -level tests and there are numerous examples of their utility (HILL et al., 1994). Most of this work has been carried out in aquatic systems but some terrestrial systems have also been used. The aquatic systems range from simple laboratory systems to complex flowing stream systems, usually referred to a mesocosms or microcosms. Microcosm and mesocosms studies with pesticides provide effect measures that are closer to the assessment measures, for several reasons (SOLOMON, 1996). Measurements of productivity in microcosms incorporate the aggregate responses of many species in each trophic level. Because organisms will likely vary widely in their sensitivity to the stressor, the overall response of the community may be quite different from the responses of individual species as measured in laboratory toxicity tests. Microcosm studies allow observation of population and community recovery from the effects of the pesticide and indirect effects of pesticides on other trophic levels. Indirect effects may result from

changes in food supply, habitat, or water quality. Microcosm studies can be designed to approximate realistic stressor exposure regimes more closely than standard laboratory single-species toxicity tests. Most studies, especially those conducted in outdoor systems, incorporate partitioning, degradation, and dissipation, important factors in determining exposure. These factors are rarely accounted for in laboratory toxicity studies, but may greatly influence the magnitude of ecological response.

Characterizing exposure

Measuring exposure in environmental matrices is one of the critical components of risk assessment but is subject to errors through improper sampling techniques and sometimes by incorrect analyses. Obtaining an unbiased and representative sample from an environmental may be very difficult and costly and yet is probably the most important part of any exposure characterization. Sampling needs to consider both temporal and spatial heterogeneity of the pesticide residues. For example, the concentration of a pesticide may vary with water depth or distance from the shore immediately after a spray-drift contamination of water. Similarly the concentration of a pesticide in flowing water may decrease over distance from the source of contamination due to breakdown in the water, adsorption to sediments, or dilution from uncontaminated water entering downstream of the source of the pesticide. Concentrations in soil may vary with the crop and soil type and with the chemical and physical properties of the pesticide as well with climatic factors such as rainfall and percolation or leaching through the soil.

The objective of sampling the environmental matrix is to obtain a characterization of exposure that will be useful in the risk assessment process. Even with a good sampling design to address spatial heterogeneity, temporal variations in concentrations may be very important in assessing risks in relation to duration of exposure and choosing the appropriate exposure time for the toxicity data. Because of hydraulic flows in a headwater stream system, peak exposure concentration may be very narrow and may be easily missed with a single daily grab sample. They would be incorporated into a continuous sampling system where daily integrated sampling was carried out but very narrow peaks would be obscured. Sampling intervals should be designed to take into account the known hydraulics and breakdown kinetics of the pesticide in question. Thus, in small headwater streams, more frequent sampling with a frequency of less than one day may be more appropriate. For slow-flowing rivers, or a rapidly degrading pesticide in a pond or reservoir, daily sampling may be adequate. For slowly degraded pesticides in stagnant pools, ponds, or reservoirs, even less frequent sampling may be needed. The concentration-time series of data that results from this type of sampling can then be analyzed by means of post processor tool such as the Risk Assessment Tool to Evaluate Duration and Recovery (RADAR) developed as part of the efforts of ECOFRAM (ECOFRAM, 1999). This tool provides information on pulse height, width and inter-pulse interval which is particularly useful for assessing likely effects on classes of organisms with known recovery times and time-exposure responses.

In many pesticide risk assessments, the actual pesticide concentrations in the environment cannot be measured and risk assessors must make use of models to predict what these concentrations will likely be. Models may be used in Monte Carlo simulations where measured or estimated distributions of input values are used to generate distributions of output values (ECOFRAM, 1999; KLAINE et al., 1996). Output from Monte Carlo simulations is useful for distributional and probabilistic analyses but, if the model is in error, the error is propagated through the entire data set. Use of Monte Carlo analysis also requires additional information on the distributions of input values, data that may not be available thus forcing the use of default or assumed values.

Several multi-compartment models for estimating pesticide concentrations in environmental media are available. The most simplified of these is GENEEC. GENEEC version 1.3 (PARKER, 1999) mimics a PRZM/EXAMS simulation of a generic 10 ha row crop field draining into a 1 ha farm pond of depth 2 m. It incorporates spray drift to estimate concentration in water at various times after a contamination event and has a choice of several crop-types. GENEEC is designed as a Tier 1 model. It is conservative and only gives one output for each use scenario. A more complex combination of EXAMS and PRZM has been used with a preprocessor called the Multiple Scenario Risk Assessment Tool (MUSCRAT) (ECOFRAM, 1999). MUSCRAT, is an application program that links chemical, crop, soil, and climate data bases and facilitates the creation of PRZM-3 and EXAMSII input files, batch processes multiple model simulations, and performs statistical analyses on predicted exposure concentrations for pesticides (ECOFRAM, 1999). MUSCRAT gives multiple values as output and these output values can be analyzed as distributions rather

than as single deterministic values. MUSCRAT is designed to provide modeled data for use in higher tiers of the probabilistic risk assessment process.

Risk assessment of pesticides

Risk assessment for pesticides is done by comparing the concentration of the pesticide estimated or found in the environmental matrix to response concentrations reported for that pesticide in the laboratory. As has been recommended numerous times, risk assessments should be conducted in a series of steps or tiers (ECOFRAM, 1999; SETAC, 1994). The use of tiered approaches in risk assessment has several advantages for the risk assessor and those being assessed. The initial use of conservative criteria allows substances that truly do not present a risk to be eliminated from the risk assessment process, thus allowing the focus of expertise to be shifted to more problematic substances. As one progresses through the tiers, the estimates of exposure and effects become more realistic as uncertainty is reduced through the acquisition of more data. Tiers are normally designed such that the lower tiers in the risk assessment are more conservative (less likely to pass a hazardous chemical) while the higher tiers are more realistic, with assumptions more closely approaching reality. Because lower tiers are designed to be protective, failing to meet the criteria for these tiers is merely an indication that a more data rich, more realistic risk assessment is needed.

Risk assessment of pesticides can be conducted for many reasons. These range from simple ranking systems to more complex probabilistic risk assessments (Figure 3).

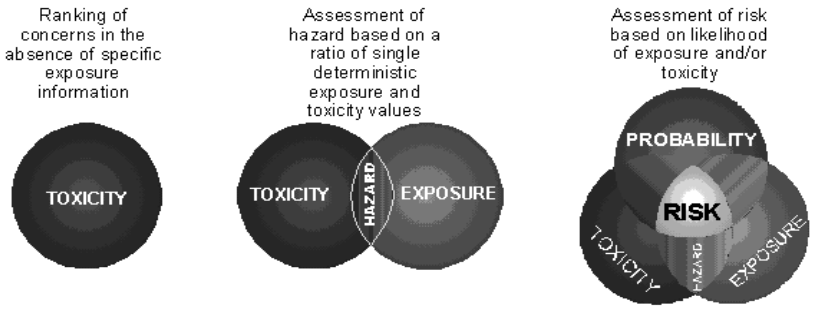

Figure 3 Illustration of the types of risk assessments applied to pesticides

The hazard quotient

The first real tier in the risk characterization process is the use of hazard quotients (Figure 3). These are simple ratios of single exposure and effects values and may be used to express hazard or relative safety. For example:

$$Hazard = \frac{Exposure\ concentration}{Effect\ concentration} \quad or\ Margin\ of\ Safety = \frac{Effect\ concentration}{Exposure\ concentration}$$

The calculation of hazard quotients has normally been conducted by utilizing the susceptibility of the most sensitive organism or group of organisms and comparing this to the greatest exposure concentration measured or estimated in the environmental matrix. This may be made more conservative by the use of an uncertainly (application) factor (CWQG, 1999) such as division of the effect concentration by a number such as 20. This is done to allow for un-quantified uncertainty in the effect and exposure estimations or measurements. In this case, if the hazard ratio is greater than 1, a hazard exists. Under the ecological risk assessment guidelines for pesticide risk assessment currently used in the US (URBAN & COOK, 1986), the

hazard ratio is referred to as the Level Of Concern (LOC) and different LOCS are used for different classes of organisms, depending on the nature of the effect measure or whether endangered species are likely to be affected (URBAN & COOK, 1986). All hazard ratio assessments incorporate some form of uncertainly factor, either explicitly as part of the calculation itself or in the criteria for acceptance of the hazard ratio. In the absence of an adequate range of toxicity tests or exposure measurements, decisions based on hazard quotients may be under protective and the use of uncertainty factors is justified. However, where an acceptable range of toxicity data are available, the inherent variation in receptor response is better defined and use of safety factors may be overprotective. The quotient approach is thus acceptable for early tiers or preliminary risk assessments but it fails to consider the range of variation that may exist in terms of exposures and susceptibility when more data are available.

Probabilistic risk assessment

The probability of occurrence of a particular event is, and has been, widely used in the characterization of risk from many physical and medical events in human society (the insurance industry) and for protection against failure in mechanical and civil engineering projects (time between failures, one-in-one-hundred-year floods, etc.). This concept has been applied in ecotoxicological risk assessment for the characterization of both exposures and effects.

Concentrations of substances in the environment can be affected by a large number of processes that relate to the amount released, the spatial and temporal distributions of the releases, and the results of the action of a large number of transportation and transformation processes (fate processes) on the substance. The likelihood and extent to which these myriad of fate processes will affect a particular quantity of substance in the environment is essentially random and frequency distributions of exposure concentrations in the environment can be used to describe and characterize the data set and to use it to make predictions similar to those made in other areas of risk management (CARRINGTON, 1996; MCBEAN and ROVERS, 1992). In many cases, these distributions fit the log-normal model reasonably well (KLAINE, et al.; 1996, SETAC, 1994; SOLOMON, 1996; SOLOMON et al., 1996; SOLOMON & CHAPPEL, 1998). The assumption of a reasonable fit to a model makes calculations of exceedence probabilities relatively easy but is not necessary for the concept of probabilistic risk assessment to be used. Centiles of distributions may be estimated from large data sets by simple ranking and interpolation or by using a suitable model, such as a polynomial, to describe the relationship. This is best used with large data sets where extrapolation beyond the observations is not needed. The same observations of log-normality generally apply to distributions of toxicological data. Many of the reactions through which toxicity mechanisms are mediated are first-order or pseudo first-order and, with a large enough data set and appropriate groupings of organisms to avoid mixing susceptible and non-susceptible species, good fits to the normal distribution are obtained (GIDDINGS et al., 2000; GIDDINGS et al., 2001; GIESY et al., 1999; HALL et al., 1999; HALL et al., 2000; HENDLEY et al., 2001; KLAINE et al., 1996; SETAC, 1994; SOLOMON, 1999; SOLOMON et al., 2000; SOLOMON et al., 2001 a; TRAVIS & HENDLEY, 2001).

Some care should be taken when using exposure or toxicity data in the distributional analyses. Exposure data should be screened to make sure that the data are consistent. Ideally, exposure data should be expressed over constant intervals such as daily samples. Distributional analysis of a data set with unequal time intervals will distort the distribution to over-represent periods where more samples were taken. In this case, samples taken more frequently can be combined as time-weighted averages. In situations where samples are taken less frequently, interpolation can be used to “generate” data with the proviso that this will introduce a bias into the data set. In temperate regions, sampling of environmental matrices may not be carried out in winter. As winter is a period of low biological activity, it may be more ecologically appropriate to focus the risk assessment on the more biologically productive months when more analytical data are available.

Effects data used in distributional analyses should be reviewed for appropriate quality (see above), however, with older pesticides more than one toxicity study may survive a quality assessment. If this is the case, several procedures may be adopted. If one of the data points represents a more sensitive life-stage and life-table analyses indicate that survival of this stage is key to population sustainability, then this data should be used. If no life-stage data are available, and/or multiple tests remain after critical review, it is recommended that test data be combined as a geometric mean as a measure of central tendency (ECOFRAM, 1999). Pesticide toxicity data reported at concentrations in excess of the maximum water solubility of the pesticide may not be reliable descriptors of responses, however, they can be used for risk assessment. These data are

almost always from the least susceptible organisms and, while less relevant in the risk assessment, can be used in the characterization of the toxicity distribution. Because the data are less reliable, they should not be used in calculating a regression line but should be included in the calculation of *n* and the plotting positions.

The toxicity and exposure data are thus analyzed as distributions on the assumption that the data represent the universe of species. Obviously, it is not possible to test all the species in the universe and, for this reason, an approximation is usually made. The same is true of exposure data as it is not practical or feasible to sample all possible locations or times. As it is unusual for sufficient toxicity and exposure data to be available to allow a cumulative frequency distribution of data to be plotted directly, an approximation to a frequency distribution is normally used (PARKHURST et al., 1996). This approximation assumes that the number of species tested (*n*) is one less than the number in the "universe". To obtain symmetrical graphical distributions (normal distributions), y-plotting positions are calculated as percentages using the formula [100 x $i/(n+1)$] (from (PARKHURST et al., 1996)) where *i* is the rank number of the datum point and *n* is the total number of data points in the set. This gives and empirical cumulative probability based on the Weibull equation. Similar empirical probabilities can also be calculated using other formulae such as the Blom equation ($P = (i - 0.375)/(n - 0.25) \times 100$) or the Hazen equation ($P = (i - 0.5)/n \times 100$) (CUNNANE, 1978). These two equations may be useful for small data sets (CUNNANE, 1978). These formulae all compensate for the size of the data set. Small (more uncertain) data sets are more likely to give more conservative estimates of high or low centiles than larger (more certain) data sets.

The principle of probabilistic approach has been described (CARDWELL et al., 1993; GIESY et al., 1999; KLAINE et al., 1996; PARKHURST et al., 1996; SOLOMON, 1996; SOLOMON et al., 1996; SOLOMON et al.,. 2000) and is illustrated diagrammatically in Figure 4. Distributional analysis can be applied to concentrations of substances in the environment with due consideration for the fact that these data are usually censored by the limits of analytical detection (Figure 4-A). In practice, all exposure concentration data below the LOD or LOQ are assigned a dummy value of zero. These data are used in the calculation of *n* but are not plotted or used to estimate centiles. As in this illustration, when plotted as a cumulative frequency distribution using a probability scale on the Y axis as a function of $\log_{10}$ concentration (Figure 4-B), these distributions approximate a straight line which can be used to estimate the likelihood that a particular concentration of the substance will be exceeded. A similar approach can be taken with susceptibility of organisms to the substance (Figure 4-C and D). The combination of these distributions in the probabilistic characterization of risk is illustrated in Figure 4-E. In this procedure, it is assumed that the distributions of sensitivity represent the range of responses that are likely to be encountered in the ecosystems where the exposures occur (SETAC, 1994). If the exposure data were collected over time at a particular site, the degree of overlap of the exposure distribution with the effects distribution can be used to estimate the joint probability of exposure and toxicity, leading to estimates of exceedence probabilities for responses at a fixed effect assessment criterion, such as, for example, the concentration equivalent to the 10th centile of the species distribution (Figure 4-E). This can be applied to a number of data sets and the resulting probabilities used for priority setting or in further assessing ecological relevance.

Expressing the results of a refined risk assessment as a distribution of values rather than a single point estimate is an approach that has been used by the Dutch Government (HEALTH COUNCIL OF THE NETHERLANDS, 1993) and recommended for use in ecological risk assessments of pesticides (ECOFRAM, 1999; SETAC, 1994). This approach has been used in a number of published and shortly to be published risk assessments of pesticides (CARDWELL et al., 1999; GIDDINGS et al., 2000; GIDDINGS et al., 2001; GIESY et al., 1999; HALL et al., 1999; HALL et al., 2000; HENDLEY et al., 2001; KLAINE et al.,. 1996; MAUND et al., 2001; SOLOMON et al., 1996; SOLOMON & CHAPPEL, 1998; SOLOMON et al., 2000, SOLOMON et al., 2001a; TRAVIS & HENDLEY, 2001; VERSTEEG et al., 1999). The major advantage of this approach is that it uses all relevant single species toxicity data and, when combined with exposure distributions, allows quantitative estimations of risks. In addition, the data may be revisited again, the decision criteria become more robust with additional data and the method is transparent (will give same results with same data sets).

The method does have some disadvantages. More data are usually needed, although these are mostly low cost studies. For new products, models have to be used to estimate exposures and models have not been widely validated for these uses. It is not easily applied to highly bioaccumulative substances where exposure is via food chain as well as matrix, however, if appropriate data are available, this can be overcome. Probably most critical of all is that it requires education of risk assessors and risk managers to increase their ability to evaluate decisions and to increase their comfort levels with the process (SOLOMON, 1996).

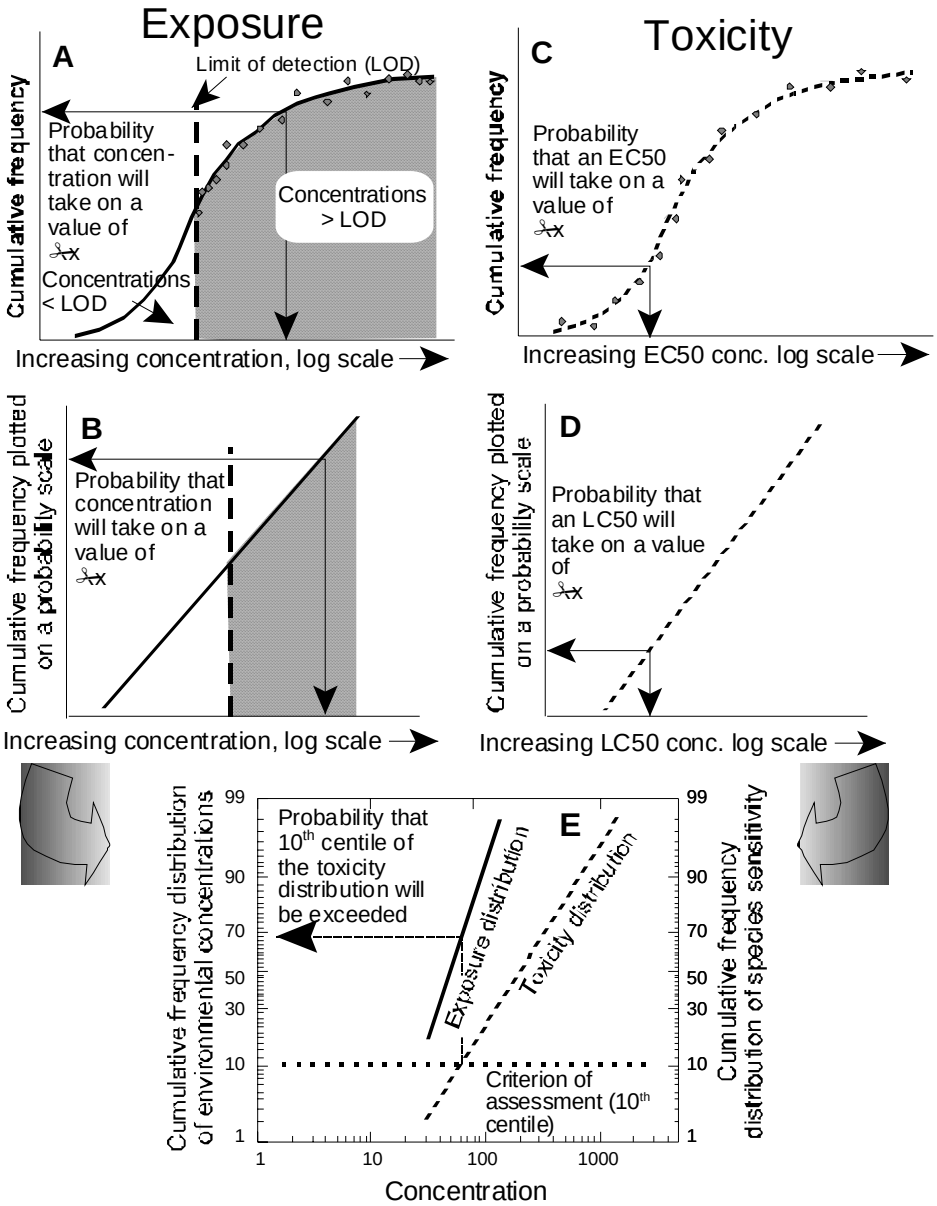

Figure 4 Illustration of the principle of the probabilistic approach (adapted from SOLOMON & CHAPPEL, 1998)

Probabilistic risk assessment can be applied to assessment based on acute or chronic responses, all that is necessary is ensure that the toxicity data and the exposure data are expressed in the same units. Although more widely used to assess risk to aquatic systems, the techniques are applicable to terrestrial systems as well (SOLOMON et al., 2001b).

Refinements to the probabilistic risk assessment process

A number of refinements to the probabilistic approach have been suggested by the ECOFRAM working group (ECOFRAM, 1999). Many pesticides have some degree of specificity in their mechanism of action. For example, herbicides may be selectively toxic to some groups of plants (weeds versus corn) as well as being less toxic to animals and other organisms that do not possess the receptor system (say photosynthesis). Similarly, an insecticide that acts on the nervous system of insects is unlikely to be highly toxic to plants. Specificity of action may not always be the case. For example, some biocides, such as the Chlorophenols, are similarly toxic to a wide range of organisms (hence their use as biocides) (LIBER et al., 1994) and the grouping of all organisms together for distributional analysis may be appropriate. Thus, from a basic understanding of the mechanism of action of a pesticide, and from the toxicity data, it may be possible to identify and group sensitive organisms that are the most likely to be adversely affected. This is helpful from the point of view of risk assessment as it allows the assessor to focus on the groups at higher direct risk and to devote less time and resources to groups exposed to very low or negligible direct risks. In addition, with a knowledge of the ecology of the potentially impacted system, it is possible to assess the likelihood that indirect effects will occur as a result of an effect on keystone groups of predator or prey/food organisms, should these be in the sensitive groups.

While the mechanism of action of the pesticide is an important criterion for grouping or organisms, habitat may also be important. For example, there may also be good mechanistic reasons to separate effects data for freshwater and saltwater organisms where it is known that one group has an inherently different sensitivity because of interactions between salinity and the pesticide of concern (HALL & ANDERSON, 1995; SOLOMON et al. 2001 a).

It is also possible to group organisms together on the basis of their reproductive strategy and life cycle. Thus, organisms which are able to recover rapidly from an adverse effect at the population level (reduction in population caused by mortality) may be considered differently from another group of organisms that may require a longer period of recovery. For example, aquatic algae have short reproductive cycles and would be expected to recover from a decrease in population more rapidly than a population of fish subjected to a similar reduction. Thus, the frequency of occurrence and the intensity of the effect that could be tolerated would be different. This is also important when deciding how the exposure data should be analyzed.

Instead of estimating the likelihood that specific toxicity criterion (say the10th centile of the species sensitivity distribution) will be exceeded, exceedence probabilities can be presented as a continuum of likelihoods. This allows the risk assessor to judge the possible adverse outcomes over a range of possible combinations in a joint probability curve. These approaches are useful for communication of risks (ECOFRAM, 1999, GIESY et al., 1999).

Using weight of evidence in ecotoxicological risk assessment of pesticides

The results of ecological risk assessments need to be interpreted in the context of a number of lines of evidence which include ecological, abiotic and biotic components of the ecosystem. While the fact that the probabilistic approach is a purely numerical methodology is an advantage from the point of view of the transparency of the procedure, it cannot, nor is it designed to, assess the ecological relevance of the exceedences that may be identified. For example, an assessment criterion of the 10th centile may include keystone organisms of value to ecosystem function. Effects on keystone species would be expected to extend to other species that are dependent on them, for example as a source of food or as a predator. For this reason, it is necessary to assess the role of the potentially affected species in terms of their function in the ecosystem and whether this can be taken over by other organisms. The probabilistic approach can be used to refine the assessment process by allowing a rational ranking of scenarios by risk (likelihood of exceeding assessment criteria) and by identifying species in the distributions for which functional redundancy may exist (less sensitive organisms that can also perform the same function as the affected organisms).

Ecological relevance can most usefully be assessed from a basic knowledge of ecology and from tests, such as microcosms, where community resiliency, productivity, and function can be evaluated directly. For this reason, refinement of the effects characterization in a probabilistic risk assessment gives a greater reduction of uncertainty.

The temporal and spatial scale of pesticide exposure is important in ecological risk assessment. The return frequency of an event (how often the event happens) is an important consideration in the choice of methods

for probabilistic risk assessment and is related to the ecological cost of recovery from the event (SOLOMON, 1996). In assessing exposure, the return frequency protected should be consistent with the resiliency of vulnerable populations. Resiliency is determined by life cycle characteristics and reproductive capacity of the potentially affected organisms and the ability of their populations (or their function in the ecosystem) to recover from the episode. The Report of the Aquatic Risk Assessment and Mitigation Dialogue Group (SETAC, 1994) recommended conservative approaches to ecological risk assessment, such as the use of low return frequencies, for example, one or fewer occurrences in thirty years. This safeguards all organisms in situations where limited information is available on mode of action or sensitivity of species. Where better information is available, more appropriate return frequencies may be used. For example, more frequent adverse events may be tolerated where a stressor affects organisms with short life cycles and high rates of reproduction. In temperate regions, many ecosystems undergo a period of dormancy and the system is, in a sense, reset seasonally by the winter. Thus, for some organisms, mechanisms for propagation beyond the winter reset already exist and resting and other dormant stages are produced from which populations in the next season will develop. Similar mechanisms exist in environments with a dry season where ephemeral water bodies are subjected to drying out. Therefore, as many organisms in these regions undergo seasonal resets, a stressor return that occurs less frequently than once per season is likely to be tolerable from the viewpoint of the long-term productivity of the population and the sustainability of function in the ecosystem, especially if the effects are spatially restricted. Protection of longer-lived species without seasonal resets, such as some fish, birds or mammals, may, however, require the consideration of return frequencies of several years.

If a stressore is present non-uniformly in the environment, unexposed areas will act as refugia (metapopulations) for repopulation of potentially impacted areas. The relative size of the exposed and unexposed areas and their closeness is important, but this issue is particularly significant for assessing risks from pesticide use, where untreated fields, set-aside land, conservation headlands, crop rotations, and mixed farming practices guarantee that refugia will be present. Similarly, refugia exist in streams and rivers and many organisms have resistant stages or propagules from which population recovery can occur. Thus, probabilistic risk assessments (and hazard quotients) are additionally conservative because they do not consider repopulation from unexposed refugia. The example of the more rapid than expected recovery of the biota in the River Rhine from an endosulfan spill illustrates this point (FRIEGE, 1986).

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Developing scenarios for estimating exposure concentrations of plant protection products in EU surface waters

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Abstract

Currently, for the EU plant protection product registration scheme (91/414/EEC), exposure concentrations in surface waters are estimated by assuming that spray drift (derived from the empirical tables of GANZELMEIER et al., 1994) enters a static, 30 cm deep water body directly adjacent to a field containing crop. It has been recognised for some time that refinement of this scenario and additional tiers are required to further develop aquatic risk assessment for agrochemicals. Over the last two years, a tiered system of 'steps' has been developed by a work group established by FOCUS (**FO**rum for **C**o-ordination of pesticide fate models and their **U**se - under the auspices of the European Commission). Step 1 is based on worst-case season loadings, and Step 2 calculates worst-case loadings based on sequential application patterns (i.e. taking account of dissipation between applications). Steps 1 and 2 are intended to be both conservative and simple to minimise further modelling for pesticides with negligible aquatic risk. Step 3 consists of up to ten scenarios that cover a representative range of locations, crops, soil types, climatic conditions, routes of potential aquatic exposure and types of water bodies. At Step 3, appropriate spray drift inputs are calculated using a spreadsheet tool based on the data of Ganzelmeier et al., and drainage or runoff inputs are estimated using the models MACRO and PRZM, respectively. Fate within the water body and exposure concentrations are modelled using TOXSWA. Modelling shells are currently being developed to facilitate the calculations for all three steps. Step 4 is less prescribed, and involves generating more localised risk assessments which can be defined from the outcome of Step 3. At this step, a range of models and geographical information may be used to further refine exposure estimates. The final report of the group will appear during 2000.

Background

FOCUS (**FO**rum for **C**o-ordination of pesticide fate models and their **US**e) was established by the European Commission in 1994 to assist in the development of environmental exposure models for the registration of plant protection products under the European Directive 91/414/EEC. FOCUS is led by a steering committee which recommends priorities for development by various working groups. Membership of the steering committee and working groups is composed of representatives from regulatory agencies, research institutes, academia and the agrochemical industry. To date, a number of working groups have discussed exposure calculations for groundwater, surface water and soil, and have produced a number of guidance documents (DOCs 4952/VI/95, 6476/VI/96 and 7617/VI/96). At the moment, two working groups are developing European scenarios for groundwater and for surface water. In this paper, the progress to date of the surface water scenario group is described. None of these proposals are yet finalised, and so the descriptions are of draft status and should be treated accordingly.

Remit of the Surface Water Scenarios Group

The FOCUS Surface Water Scenarios Group was established in 1997. It was charged with developing a set of standardised modelling scenarios for drift, drainage and runoff entry routes into surface water. The scenarios should be based on a tiered sequence of exposure assessment *steps*, namely:

- Step 1 = Worst-case loadings
- Step 2 = Worst-case loadings based on sequential application patterns (i.e. taking account of dissipation between applications)
- Step 3 = Realistic worst-case based on crop/climate scenarios (using realistic worst-case soils, topography, water bodies, climate, agronomy)
- Step 4 = Localised/regionalised risk assessment, moving away from edge-of-field to the landscape level.

Development of Step 1 and 2 scenarios

Step 1 and 2 are, by design, very conservative estimates of predicted environmental concentration in surface water (PECsw), based on the total season pesticide loadings. The purpose of Steps 1 and 2 is to identify plant protection products which have large safety margins (i.e. they present negligible risks to aquatic ecosystems). Unlike the current EU procedure which only includes spray drift inputs, a runoff/drainage component is also included at Steps 1 and 2 to account for additional potential exposure for those compounds which might leach through field drains or run off into surface waters. Calculations at Step 1 and 2 are relatively simple and are performed using a spreadsheet. The outputs from this spreadsheet include the initial PECsw, various concentrations through time and time-weighted averages (TWA) appropriate to the effects endpoints for risk assessment (acute and chronic exposure durations). The scenarios were developed using combination of existing precedent and 'expert judgement'. It is intended to perform a check on assumptions selected for Steps 1 and 2 by eventually comparing the selected pesticide input parameters with those developed at the more empirical step 3 (see below).

At Step 1, a static, shallow water body is assumed to be exposed. The water body contains water of 30 cm depth and a 5 cm depth of sediment (bulk density of 0.8 g/cm^3 and organic carbon content of 5%). Inputs are based on the maximum annual application rate, unless the dissipation rate (DT50) of the compound in water-sediment studies is less than one third of the application interval. In this instance, inputs are based on a single application because such rapid dissipation will not lead to accumulation of the compound in the aquatic environment.

Pesticide loadings at Step 1 include spray drift ranging from 3-26% drift (depending on the crop type). These values are derived from the spray drift data of GANZELMEIER et al. (1994), and the value selected is the 90th percentile of these data. The 90th percentile was selected to reduce the compounding of worst-case upon worst-case, which potentially results in a very extreme scenario. At step 1 it is assumed that there is an additional input of 10% of the application rate from a 'drainage/runoff' component. This value was considered by the group to be a suitable worst case, based on expert judgement. Overall therefore the total amount of input at Step 1 can range from 13 to 36% of the application rate. Dissipation of the pesticide at step 1 is calculated on the basis of data from laboratory water-sediment studies. Once the pesticide enters the water body, the dissipation process is assumed to begin on 'day 2' (i.e., the initial PEC does not account for any adsorption).

Calculations performed at Step 2 are the same as step 1 except that:

- Applications are treated individually with realistic intervals between treatments
- Each application has a separate drift component which enters on day of treatment - combined probability of 90%
- Each application subject to soil degradation, then 4 days after last application residue is lost from soil via runoff or drainage (depending on location/season of use - from look-up tables)

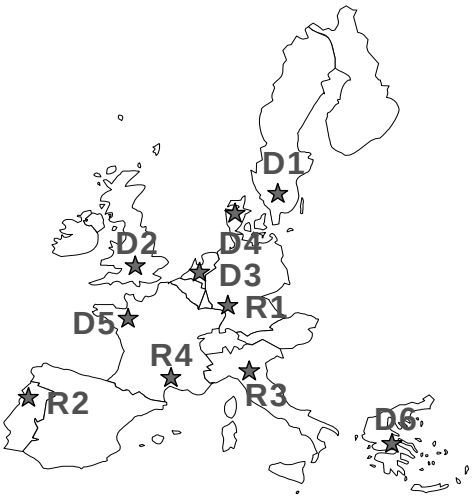
Development of Step 3 scenarios

The aim of the group was to produce a maximum of ten realistic worst-case surface water scenarios that are broadly representative of European agriculture, and include reasonable combinations of drift and drainage or runoff. Where possible, it was felt that it would be useful to identify existing monitoring sites in the scenarios to allow subsequent validation

In order to develop the scenarios (Figure 1), EU data layers on climate, slope, landuse and cropping were collected. The main drainage areas (scenario number prefixed by D) were identified by recharge capacity, and areas prone to runoff (scenario number prefixed by R) were selected based on spring daily rainfall. For each of the areas, appropriate soil types and slopes were derived. For each scenario, appropriate crop types were identified.

It was considered that for agricultural landscapes, and considering the edge-of-field scale of the risk assessment at Step 3, appropriate water bodies would be streams, ditches and ponds (depth x width x length: on average about 0.5 x 2.0 x 100 m, 0.3 x 2.0 x 100 m and 1.0 x 30 x 30 m, respectively) with hydraulic residence times of about 0.1, 5 and 50 d, respectively. The water bodies for each scenario were selected

based on expert judgement, and a preliminary evaluation of these selections was made by consulting topographic maps for the scenarios and identifying the water bodies present.



D1. Scandinavia, clay, cold, moderate precipitation, gently sloped, drainage scenario (weather station: Lanna)
D2. North-west Europe, clay, temperate, moderate precipitation, gently sloped drainage scenario (weather station: Brimstone)
D3. Northern maritime, sand, temperate, moderate precipitation, flat, drainage scenario (weather station: Vredepeel)
D4. Northern maritime, loamy, temperate, moderate precipitation, gently sloped, drainage scenario (weather station: Skousbo)
D5. Western maritime, heavy loam, temperate, wet, moderately sloped, drainage scenario (weather station: La Jailliere)
D6. Eastern mediterranean, heavy loamy, warm, moderate precipitation, gently sloped, drainage scenario (weather station: Thebes)
R1. Central european, silty, temperate, wet, gently sloped, runoff scenario (weather station: Weiherbach)
R2. Atlantic, southern maritime, loamy, temperate, very wet, very steep, runoff scenario (weather station: Porto)
R3. Central european/ mediterranean, sandy loamy, warm, wet, steep, runoff scenario (weather station: Bologna)
R4. Southern european/ mediterranean, loamy, warm, moderate precipitation, moderately sloped, runoff scenario (weather station: Roujan)

Figure Step 3 Soil-Climate Scenarios (D prefix denotes a drainage scenario and R a runoff scenario)

Step 3 deterministic modelling

At Step 3, the modelling of exposure concentrations follows the actual sequence of applications. A number of scenario parameters have been developed to enable to the modelling to be conducted, including:

- typical weather over a 16 month window;
- soil profile (texture, organic carbon, bulk density, moisture holding capacity, conductivity, parameterization of macropores);
- crop data (planting date, lay, harvest date, etc.);
- topography and hydrology of water bodies.

Chemical inputs into and fate in the water body are calculated as follows:

- Drift - a ‘drift calculator’ has been developed based on the data of Ganzelmeier *et al*., which integrates drift over water body dimensions, and adjust the drift percentile according to the number of applications, resulting in an overall 90th percentile;
- Drainage - calculated using the model MACRO;
- Runoff - calculated using the model PRZM3.
- Fate of the chemical inputs in the various water bodies are calculated using the surface water model TOXSWA, resulting in a range of chemical concentrations over time.

Step 4 principles

Step 4 represents a higher tier of exposure modelling, and as such is far less prescriptive than the earlier steps. The approach at step 4 will be to move away from the edge-of-field, and consider exposures at the broader, landscape scale. The intent of step 4 is to allow any potential risks which have been identified in a step 3 risk assessments to be further refined, and consequently this will result in a localised or regionalised risk assessment focused on those areas or water bodies which have been identified as being at risk.

modelling requirements at step 4 will therefore need to be defined on a case-by-case basis. Step 4 modelling will also allow various mitigation options to be explored.

The sorts of considerations which could be used to refine the modelling at step 4 could include information on:

- product use data (how much is used, what is the market penetration),
- realistic landscape characteristics (e.g. from remote sensed data or geographical information systems)
- local weather over longer periods, etc.

Surface Water Scenarios Group outputs

The progress described above summarises the work to date. As yet, none of these proposals has been finalised, and so the discussion should be considered as a draft and treated accordingly. A final report will be produced by the group during 2000, and it is also intended to develop software shells for steps 1, 2 and 3. These will be available from FOCUS in due course.

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Does risk mitigation need also modified approaches in toxicity testing?

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Abstract

It is demonstrated by an example from mesocosm experiments and one from the *Daphnia* reproduction test, that the representativeness of *Daphnia* for the community, where it be logs to, is limited, which can lead to an underestimation of the community sensitivity. Additionally, the *Daphnia* reproduction test can overlook toxic effects because the offspring produced during the test is not assessed further with respect to survival and reproduction. Therefore, we recommend (1) to select test species according to their taxonomic rather than to their trophic position, and (2) to switch to testing of populations, which assesses also effects in subsequent generations.

Introduction

Although risk mitigation mainly deals with measures to reduce the risk potential of already authorised substances to non-target field populations, living in habitats adjacent to sprayed agricultural crops, we wish to explain that risk mitigation should already start much earlier and could be enhanced with a modification of the current approach in environmental toxicity testing. According to the current philosophy, in lower tier testing representative species of the trophic levels are subjected to toxicity testing and the results are extrapolated to other members of the same trophic level (OECD, 1993; FENT, 1998; SHAW & CHATWICK; 1999). We will show by our first example from outdoor aquatic mesocosm tests that this philosophy could be dangerous and non-protective to the community, if the current risk assessment practices are applied.

Another problem with current toxicity testing arises from the practice that the experimental test cohorts are rarely examined for a sufficient time period and mostly with only one generation. Hence, the response of sensitive instars and effects on subsequent generations cannot be observed and incorporated into risk assessments. In our second example we compare results from *Daphnia* reproduction tests (OECD, 1997) with those obtained in population experiments, conducted with the same species under corresponding treatment and environmental conditions.

The current approach and its limitations

Example 1: Sensitivity differences between herbivores of the same community

Table 1 presents the EC_{50} of some herbivores from two outdoor mesocosm experiments conducted in 1998 and 1999, to study the fate and effect of two fungicides (because the fungicides are just in the process of authorisation the names have yet to be kept secret). In both experiments, a spray application method was chosen to simulate the entry of the test substance into a water body by direct overspray or spray drift. Eight (1998) and four (1999) separate, identical applications of the test substance were carried out in each pond with a time interval of 7 days between applications, which started in June (1998) and July (1999) and lasted until August. The test concentrations were verified by residue analysis.

The results of Table 1 are based on the aforementioned treatment periods, for which the mean abundance of eight and four sampling dates, respectively, was determined per treatment. Then, a 4-parameter logistic concentration/response function was fitted to the concentration/abundance relationship using a non-linear-regression approach, for which the statistical significance is also given in Table 1.

We found that *Daphnia magna*, the ”backbone” of testing trophic level 2 in aquatic systems, was less affected by the fungicides and showed a response of about the same magnitude for both fungicides. Copepod nauplii proved to be about one order of magnitude more sensitive.

However, tremendous sensitivity differences between *Daphnia magna* and some rotifers were observed. With Fungicide 1, the EC_{50} of the most sensitive rotifer (*Brachionus* spec.) was nearly two orders of magnitude lower than in *Daphnia magna*, whereas with Fungicide 2 the difference between the highly sensitive *Keratella quadrata* and *Daphnia magna* was even about three orders of magnitude.

Table 1 EC_{50} of selected herbivores from an outdoor mesocosm community, obtained for two fungicides; the EC_{50} is based on the mean abundance during treatment period; R^2: coefficient of determination of a non-linear regression using a 4-parameter logistic function; *, **, ***: significance indicators for $p \leq 0.05$; $p \leq 0.01$ and $p \leq 0.001$, respectively

	Fungicide 1		**Fungicide 2**	
Taxon	**EC_{50} [µg/L]**	**R^2**	**EC_{50} [µg/L]**	**R^2**
Daphnia magna	407.6	0.827	327.9	0.841 *
Copepod nauplii	56.5	0.941 **	29.3	0.911 *
Sum of rotifers	11.9	0.980 ***	1.0	0.972 **
Keratella quadrata	26.8	0.913 *	0.1	0.986 ***
Brachionus spec.	9.2	0.954 **	42.5	0.905 *

Example 2: The daphnia reproduction test does not tell the complete story

In the second example we present results from a study, in which the dispersant "Dispersogen A" was used in the *Daphnia* reproduction test and in a population experiment. Dispersogen A is a condensation product of naphtalenesulfonic acid with formaldehyde; it is commonly used in dye production and in the textile dying process of insoluble dyes. More background information, experimental details, and complete results have been recently published (HAMMERS-WIRTZ & RATTE, 2000; for details see there).

Table 2 gives the EC_{50}, obtained for various parameters of two *Daphnia* reproduction tests, one F1 reproduction test, and one population experiment. In both *Daphnia* reproduction tests, the mortality was found to be increased with increasing dispersant concentration. At the same time, up to a dispersant concentration of 10.2 mg/L the daphnids produced up to 53% more living offspring than in the control. Only at 25.6 mg/L, less offspring was released compared to control. Also with increasing dispersant concentrations, the neonates showed higher mortality and were found to be smaller that those of the control.

Table 2 Overview over the EC50, obtained for various parameters of *Daphnia magna* in three reproduction tests and one population experiment; note that in the F1 test the young daphnids introduced stemmed from mothers grown in dispersant treatments but were grown in uncontaminated medium; here the potential F2 offspring number was calculated with F0 offspring number, F1 offspring number and F1 survival

	EC50 [mg/L]
Reproduction Test 1	
Mortality	16.5
Total offspring number	10.2 < EC50 < 25.6
Reproduction Test 2	
Mortality	> 10.2
Total offspring number	> 10.2
Neonate body size	> 10.2
F1 Reproduction test	
Total offspring number	0.53
Potential number of F2 per F0 female	0.1 < EC50 < 1.64
Population experiment	
Average population size	2.15
Carrying capacity	4.14
Growth rate	3.56
Neonate mortality	7.00

Neonates from the second reproduction test were themselves introduced as young ones into a F1 reproduction test without any further dispersant treatment – only their mothers had been exposed to the dispersant. The daphnids born in the dispersant treatments remained significantly smaller than the controls

throughout the test period. All daphnids born in the dispersant treatment produced up to 70% less offspring than the controls. At the same time the mortality of this offspring increased with increasing exposure concentrations of the mothers.

The population experiment revealed that the control showed the fastest population growth, where as growth became slower with increasing dispersant concentration. Also the average population density decreased with increasing dispersant concentration. Table 2 gives a summary of the main effects using the EC_{50} as toxicity measure. It reveals that the population parameters were about one order of magnitude more sensitive than those in the *Daphnia* reproduction test.

Discussion

We have shown that due to the current selection of test species and to the manner, how the test procedure is designed, the outcome of a toxicity test can lead to sensitivity estimates, deviating much from the real sensitivity of a community, for which risk assessment is performed (Example 1). Also the sensitivity of a single population or species can be underestimated (Example 2). This means that risk assessment can loose its protectiveness, if it is based on *Daphnia magna* as the alone representative (or - generally spoken - on too few representatives) of the herbivorous trophic compartment in stagnant freshwater systems.

The first example clearly supports a change from the trophic approach, i.e. choosing the test organisms according to the trophic level, to the taxonomic approach. The latter means that the selection of test species should be done such that enough representatives from the dominant species of a community are included in the test battery. The reasoning behind is that the sensitivity of a species depends on its physiological properties, which are more related to the taxonomic rather than the trophic position of a species. Although the shortcoming of this approach has been already recognised by environmental toxicologists (see, e.g., SHAW & CHATWICK, 1999), by the time being the trophic approach is favoured with current toxicity testing. Advocates of the trophic approach see only problems with extreme cases, in which *Daphnia* would stand as a representative of, e.g., the hippopotamus (also a trophic level 2 species) rather than for the normal case, in which the species belong to the same community, as considered here. Our example clearly shows, that the trophic approach can be careless and dangerous, since the sensitivity of a community cannot always be correctly and confidently estimated.

Also the design of toxicity tests needs to be reconsidered – in particular with respect to measuring sufficient variables of importance for the population, which is the assessment end point. Even in the *Daphnia* reproduction test, which is commonly seen as a real life-cycle test, highly appropriate to extrapolate to the population level (see, e.g. RATTE, 1996), important effects on the next generation are unnoticed, since the quality of offspring produced by the treated mothers is generally not assessed. In our example, the daphnids produced more offspring when treated with the dispersant, and the neonates were visibly smaller, which leads to additional testing. In many cases of routine testing, the neonates could appear quite normal for the experimenter, although they could be affected during the embryonic development and this damage could evoke mortality and inhibition of reproduction in the subsequent life span.

The examples presented here were found by chance and no systematic search for this type of bias in sensitivity estimates was undertaken. If we take the combination of results of only the two examples presented as the worst case, the outcome of the Daphnia test could lead to toxicity measures (e.g. EC_{50}) of about four orders of magnitude too high (= too insensitive) relative to the sensitivity of the community. The current risk assessment involves predicting the amount of a substance that will enter the environment and comparing this with results from acute and prolonged toxicity studies. In order to account for uncertainties the concentration of environmental concern is computed from the results (LC_{50}, EC_{50}, NOEC) of most sensitive species from toxicity tests divided by a safety factor. According to OECD (1992) a safety factor of 1000 is used, if data from one or two acute tests are available, it is 100, in case the acute LC_{50} or EC_{50} are available for the base set (alga, daphnid, fish) and 10, if the chronic NOEC is available for the complete base set. A comparison of these safety factors and the orders of magnitude, by which sensitivity could be underestimated, clearly shows that this magnitude of difference cannot be balanced even by the highest safety factor.

Toxicity testing of pesticides must result in measures, which under the current risk assessment practice ensures a sufficient protection of natural field communities. This however requires a test strategy, which not only protects a population of a distinct species, seen as important from aspects of nature/species conservation

and some ecosystem function. It is the natural community, e.g. of a small stream or shallow pond, as a whole that has to be protected. It is comprised of many organisms from different taxonomic and trophic levels. Protection the structure of a community ensures automatically also the protection of its function.

We therefore recommend to include more (not all) species from different taxonomic groups, which play a major role in the considered community, into toxicity testing, such as representatives from rotifers, molluscs, oligochaetes, insects, etc. In addition we plead for replacing current single-species toxicity tests by population experiments, whenever possible.

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Regulatory tools

Buffer zones to protect aquatic life from pesticide spray drift, and development of the 'LERAP' approach

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Abstract

Buffer zones are used in the UK to protect aquatic life from spray drift. Prior to 1999, a buffer zone for a pesticide product applied via a boom sprayer was fixed at 6 m. However, a need was identified for a more flexible, practical, and enforceable approach. Hence, the Local Environmental Risk Assessment for Pesticides (LERAP) Scheme has been developed, which has the aim of allowing flexibility in buffer zone distance whilst maintaining a high level of environmental protection. Since the scheme was launched in March 1999, there has been a positive reaction from growers. A similar scheme for applications by broadcast air-assisted sprayers to hops and orchard crops is currently being developed.

Introduction

'Buffer zones' (also known as 'no spray zones') have been used in the UK since 1990 to protect aquatic life from spray drift.

A standard distance of 6 m (from sprayed area to water surface) is used where application is by tractor mounted arable sprayers (i.e. boom sprayers). This distance was intended to represent the length of a typical spray boom section, which could be turned off during application in order to comply with a buffer zone. (NB This standard distance has now been changed to 5 m from sprayed area to the top of the bank of a water-body; see paragraph on 'Reference points').

For applications to orchards by broadcast air assisted sprayers buffer zones are also used. The width of these is not standard, with the distance being set according to the result of the risk assessment. The most typical distances which have been used for orchards are 15 and 18 m, but buffer zones as narrow as 5 m and as wide as 38 m have also been imposed. Some products applied to hops have also attracted buffer zones. As in the orchard situation, the distance used can vary according to the result of the risk assessment.

A standard 2 m buffer zone (between the sprayed area and the water surface) is also implemented where applications are by hand-held sprayers. (NB This standard distance has now been adjusted to 1 m between the sprayed area and the top of the bank of the water-body; see paragraph on 'Reference points').

It should be clarified that the term 'buffer zone' or 'buffer strip' can also be used to describe vegetated borders at field edges designed to minimise pesticide runoff. In the context of this paper a buffer zone is only intended to minimise spray drift, and may be cropped or uncropped.

How are aquatic buffer zones triggered?

Buffer zones are set on a product-by-product (or active substance) basis as a result of specific risk assessments. Currently, approximately 380 products in the UK carry a standard 5 m buffer zone.

Risk assessments are conducted by utilising laboratory acute and chronic toxicity data, and comparing the results with predicted environmental concentrations (PEC) in surface water. One of the major input routes taken into account is spray drift. Spray drift deposition data in conjunction with a model 30 cm deep static water body are used to derive PEC values. Resulting toxicity exposure ratios (TERs) are then compared with the relevant Annex VI 91/414 EEC trigger values. If the trigger is breached, the risk assessment is refined for example by the use of fate and behaviour data, if appropriate. If the triggers are not met by the refined assessment then a buffer zone is imposed (if appropriate higher tier data are not available). Subsequently, companies can submit further data such as from mesocosm or microcosm studies in order to remove a buffer zone restriction. After a thorough assessment of new data, buffer zones have been removed on a number of occasions.

Legal status of buffer zones

Buffer zones are statutory, and are clearly stated on product labels. Hence, it is an offence for a user to apply a product without complying with a buffer zone restriction. The responsibility for enforcing buffer zones at the farm level lies with the UK Health and Safety Executive (HSE). Officers from this body carry out routine inspections of farming premises to ensure that, for example, pesticides are being stored correctly and that records of pesticide applications are up to date. Enforcement of buffer zones is a difficult issue. Prior to the introduction of the LERAP scheme, action could only be taken if an officer from the HSE happened to observe a farmer actually spraying without adhering to a buffer zone, where one was necessary for the product (or products) being applied. This relied not only on the officer being in a specific field at exactly the right time, but also on them having knowledge of what was being sprayed.

The need for a review of buffer zone policy

Those in the farming industry complained about a lack of flexibility, which they said made buffer zones impractical in the context of agricultural production. There were also problems with enforcement as discussed above.

Over the last few years much work and discussion has been put into reviewing the use of buffer zones. The two key elements considered in this review were to increase flexibility and improve enforceability.

The underlying principle behind the first element is that if some flexibility is introduced, the level of protection of aquatic life must be maintained. Discussions led to the conclusion that local factors should be taken into account when setting appropriate buffer zone distances, and that if this was to be achieved, some kind of risk assessment at the farm level was needed. Hence, the idea of Local Environmental Risk Assessment for Pesticides (LERAP) was born. Instead of complying with a fixed buffer zone distance as specified on the product label, the user would be free to carry out a local assessment to determine whether the stated buffer zone could be reduced.

Local Environmental Risk Assessments for Pesticides Scheme (LERAPS)

It was recognised from a very early stage that if the LERAP scheme was to be introduced then it must be straight forward and practical to use, otherwise there would be no improvement in compliance and, hence, environmental protection. Clearly, the scheme would be statutory and it was proposed would include the requirement for records to be made of each of the LERAP assessments undertaken. The requirement to maintain written records could aid enforcement, as these would provide tangible evidence that a LERAP had or had not been performed. Hence, legal actions could be taken if the appropriate records are not made.

Initially, work was required to determine which local factors could be taken into account and how these might influence the buffer zone distance necessary. Discussions, including at the UK's Advisory Committee on Pesticides (ACP), recognised that factors could be divided between those which are measured or observed, and those which are under the direct control of the farmer or spray operator. Factors considered under the measured or observed category were wind speed and direction, flow rate of water course, size of watercourse, ecological 'quality' of water-body. Factors under the farmers control were application rate, use of wind breaks, and spray equipment.

The ACP considered which factors it was appropriate or practicable to take into account. It was considered that wind speed/direction was not an appropriate factor to use as this may change during the course of a single spraying operation. There were also practical problems with measuring wind speed accurately and whether measurements should be taken during the spraying operation. Flow rate of the watercourse was also discounted, as again, there were problems with the practicality of taking measurements. The ecological quality of the surface water to be protected was also considered. However, it was contended that it was not reasonable to assume that a pesticide user has the expertise to classify a water-body. Hence, the 'quality' factor was also excluded from the LERAP scheme. Using size (width) of a water-body, taking account of the potential for dilution, was thought to be viable. Similarly, it was concluded that reductions in application rates might allow reduced buffer zone widths.

It was recognised that, in the long term, new sprayer technology and engineering controls may offer the most significant contribution to reducing drift and hence buffer zone widths. It was agreed that this aspect should be incorporated into the LERAP scheme. It was recognised that there was a need for independently validated data to clearly demonstrate the drift reductions afforded by specific sprayer 'set ups' or pieces of equipment.

Hence, the conclusion was that application rate, water-body size, and engineering controls could be used in the LERAP scheme. This interim recommendation was agreed by Government Ministers. At this stage there was a public consultation exercise to gain feedback from interested parties including pesticide users and bodies representing environmental issues. The LERAP concept was generally well received and it was decided to pursue this option. The next task was to determine how changes in application rate, water-body size, and spray equipment might allow commensurate reductions in buffer zone width. It was decided to first tackle the issue of the 6 m arable buffer zone, before developing approaches for applications to hops and orchards. The rules were developed and were agreed by the ACP and were also subject to a second public consultation exercise. Government ministers agreed to the launch of the scheme, including the rules set out below.

Deriving the rules for the arable LERAP scheme

The following rules were derived on the basis that the standard buffer zone is 6 m from the directly sprayed area to the water surface. The rules are only relevant to application by vehicle mounted or drawn downward placement sprayers (i.e. boom sprayers).

Application rate

The spray drift data currently utilised are those produced by GANZELMEIER et al (1995). These data indicate that drift deposition at 5 m is 0.6% (95th-percentile). This value is used to assess spray drift from use with a buffer zone in place (there is no specific value available for drift a *6* m). The spray drift deposition at 3 m from the area of direct spray is 1.0% (95th-percentile). Hence, the amount of deposition at 6 m is approximately half that at 3 m.

If the application rate is halved, the deposition of that product in terms of drift at a specified distance will also be halved compared with an application at the full rate (all other factors being equal). Therefore, if a product is applied at half rate, the deposition of product in terms of spray drift at 3 m should be equivalent to the deposition at 6 m resulting from a full rate application. Overall, it is possible to derive a simple rule, i.e. if the rate is halved the buffer zone distance can also be halved.

Engineering controls to reduce spray drift

New technology in pesticide spray application offers benefits in terms of reduced drift. Independently assessed data on the performance of sprayers (or more accurately, sprayer configurations) were needed. It was proposed that an accreditation scheme would be set up for sprayer performance. Drift levels could be compared with a ‘standard’ reference sprayer, and the equipment under test may be allocated a *, ** or *** ‘LERAP low-drift’ rating. A protocol for testing was produced under which the sprayers could be assessed. It is intended that equipment manufacturers would generate the data in order to achieve a LERAP low drift status. The procedures for the submission and assessment of data on low drift equipment have been put in place.

Size of water body

The use of this factor has been developed by the UK Environment Agency. Based on the assumption that larger (wider) water bodies give a greater capacity for dilution of the pesticide loading, water bodies which are greater than 3 m in width will enable the standard 6 m buffer zone to be reduced. Specifically, water bodies 3 to 6 m in width would need a 4 m buffer zone. Those greater than 6m would have a 3 m buffer zone.

Dry ditches

The current definition of surface waters to be protected includes ditches which are dry at the time of application. Hence, the 6 m buffer zone applied to dry ditches. It was agreed under the LERAP scheme that dry ditches would have a buffer zone of 2 m (or 1 m using the revised reference points; see paragraph below). This reduced buffer zone is relevant for products in both Categories A and B (see below).

Combining the factors

Under the LERAP the factors described above can be combined. The appropriate buffer zone widths for combinations of factors have been set out in matrices (see Annex A).

Which pesticides can be subject to the LERAP approach?

It was agreed that for some products with buffer zones the LERAP approach would not be permitted, until it is demonstrated that a sufficient level of protection of aquatic life is provided (through a reassessment of risk to aquatic life). The products which are not included in the scheme are those which contain organophosphorus insecticides or synthetic pyrethroids. These products were put in 'Category A'. Other products with buffer zones are included in the scheme, and these were placed in 'Category B'.

Reference points

Previously, buffer zones were set with the reference points being the edge of the direct sprayed area and the edge of the water surface of the water body being protected. Given the variation in water levels over a season, the latter reference point could move over the year. For practical and enforcement reasons it was considered better to have a reference point which was fixed. This was deemed to be the top of the bank. In order to take account of the fact that the reference point was further away from the water surface, the standard buffer zone was adjusted to 5 m. The assumption is that the previous level of protection is maintained. All relevant approval holders were instructed to amend their product labels to take account of this change.

Example compounds

Organizers of the workshop have asked for details of how lambda-cyhalothrin and glyphosate trimesium are covered by the risk mitigation measures discussed. Given that the first compound is a synthetic pyrethroid it is not eligible for reduced buffer zones under the LERAP scheme. The second compound does not have a buffer zone. Both compounds are currently being reviewed under Directive 91/414 EEC. In due course this will involve a reassessment of the need for appropriate risk mitigation measures at Member State level.

Implementation of the ARABLE/LERAP scheme

Activities since the launch

Perhaps the implementation aspect of the scheme offers the greatest challenge. The ARABLE/LERAP scheme was launched in March 1999. A booklet (MAFF Publications, 1999) on the scheme, together with the list of Category A and B products, was sent to all arable farmers in the UK and many other interested organizations and individuals. So far around 130,000 copies have been dispatched. A web-site was also set up to provide information on LERAPS and an up-to-date list of products in Categories A and B. In the first few weeks after the launch the Pesticides Safety Directorate answered hundreds of telephone enquiries on all aspects of the scheme. Many callers were keen to know what low drift equipment was on the *, **, and *** lists. Low drift nozzles (when used under specific pressures and other conditions) from two companies have already been assessed and included in the *** list, which means that a 1 m buffer zone can be used. As more equipment becomes officially recognized, the value of the scheme to growers will increase significantly.

Other publicity of the scheme has included wide coverage in magazines such as Crops and Farmers Weekly, and also exhibits at agricultural events such as Cereals '99. The agricultural consultants ADAS have assisted in getting the LERAP message to pesticide users at the farm level.

Reactions of farmers

The general reaction of farmers has been positive and awareness is high. The most notable benefits have been increased awareness of the risks to watercourses and where they are on farms. Many farms now have maps of watercourses for the first time. The most notable problems arise for smaller farmers. However, this difficulty seems to arise mainly from the general increase in legislation being introduced affecting farmers, of which the LERAP scheme is only one part. For those who find the scheme too complex the option still exists for them to simply comply with the standard 5 m buffer zone.

Future actions

Meetings are being held with software companies on how the scheme can be incorporated into future editions of farm management software packages. Further publicity is planned, for example to coincide with autumn spraying activity. Training providers will be covering the LERAP scheme in courses for advisers and pesticide users. Discussions are also underway on a compliance monitoring exercise, to judge how well the scheme is being adopted at the farm level.

Development of a LERAP scheme for orchards and hops

The need for a LERAP scheme in orchards and hops has also been highlighted. Orchards are a source of high value crops where marketability is dependant on fruit size and quality. Any factors (such as leaving wide untreated buffer zones) that may adversely affect pest and disease control can impact on these parameters. Similarly in hops high pest and disease pressures could result in crop losses in untreated areas. While the area grown is relative small compared with the area of arable crops, broadcast air assisted sprayers used to treat orchards and hops can generate substantial spray drift. The prospect of better compliance and enforcement under a LERAP scheme could therefore make a worthwhile contribution to better environmental protection.

Rules for reduced application rates and low drift machinery, leading to reduced buffer zones, are currently under discussion. Further information on influence on windbreaks on spray drift is needed in order to investigate whether this factor can also be taken into account.

Conclusions

The LERAP scheme provides a means of maintaining a high level of environmental protection whilst reducing agronomic impact compared with fixed buffer zones. The requirement for written records increases potential for enforcement. The aim is to improve compliance, and to generate greater awareness of the importance of protecting surface waters. In the first six months since the launch of the arable scheme a positive response has been received from growers.

Annex

Combining the factors for the ARABLE/LERAP scheme

N.B. The distances specified are set on the basis of the previous reference point of the edge of the water surface. For the new reference point (top of the bank) distances should be reduced by 1 m.

Table A Standard reference sprayer

Application Rate **Size of water body**	**Full Rate** **(75.1 - 100%)**	**¾ Rate** **(50.1 – 75%)**	**½ Rate** **(25.1 - 50%)**	**¼ Rate** **(0 - 25%)**
All water bodies less than 3 m	6 m	5 m	3 m	2 m
All water bodies 3 - 6 m	4 m	3 m	2 m	2 m
All water bodies 6 m or wider	3 m	2 m	2 m	2 m
Dry ditch connected to river system	2 m	2 m	2 m	2 m

Table B LERAP - low drift* sprayer

Application Rate **Size of water body**	**Full Rate** **(75.1 - 100%)**	**¾ Rate** **(50.1 - 75%)**	**½ Rate** **(25.1 - 50%)**	**¼ Rate** **(0 - 25%)**
All water bodies less than 3 m	5 m	3 m	2 m	2 m
All water bodies 3 - 6 m	3 m	2 m	2 m	2 m
All water bodies 6 m or wider	2 m	2 m	2 m	2 m
Dry ditch connected to river system	2 m	2 m	2 m	2 m

Table C LERAP - low drift** sprayer

Application Rate **Size of water body**	**Full Rate** **(75.1 - 100%)**	**¾ Rate** **(50.1 - 75%)**	**½ Rate** **(25.1 - 50%)**	**¼ Rate** **(0 - 25%)**
All water bodies less than 3 m	3 m	3 m	2 m	2 m
All water bodies 3 - 6 m	2 m	2 m	2 m	2 m
All water bodies 6 m or wider	2 m	2 m	2 m	2 m
Dry ditch connected to river system	2 m	2 m	2 m	2 m

Table D LERAP - low drift*** sprayer

Application Rate **Size of water body**	**75.1 - 100%**	**50.1 - 75%**	**25.1 - 50%**	**0 - 25%**
All water bodies less than 3 m	2 m	2 m	2 m	2 m
All water bodies 3 - 6 m	2 m	2 m	2 m	2 m
All water bodies 6 m or wider	2 m	2 m	2 m	2 m
Dry ditch connected to river system	2 m	2 m	2 m	2 m

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Realistic exposure scenarios for the contamination of surface waters – a GIS approach

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Abstract

Today, more and more environmental information is available in a digital form. This offers the opportunity to use Geographical Information Systems (GIS) to introduce local environmental conditions into risk mitigation concepts for the use of pesticides. Depending on the type of pesticide, the spraying amount and the application technique on one hand, the size of the surface water, the type of bank vegetation and other environmental information can be taken into account on the other hand. On this basis risk categories can be defined on a local scale and the maximum distances can be reduced where exactly defined environmental triggers are fulfilled. This concept would offer more possibilities for agricultural practice concerning the use of pesticides and for environmental protection on a local scale. In this context GIS would be a helpful tool for managing such a concept and to ensure an easy handling of the information neccessary.

Introduction

Risk evaluation of surface water contamination with pesticides by drift and runoff is based on a mathematical evaluation of the following parameters today:

- stagnant water with a depth of 30 cm and a width of 1 m
- no vegetation between the application area and the surface water
- equal pesticide entry in the whole water body
- existing communities of algae, invertebrate animals, and higher water plants
- application of the maximum pesticide amount allowed.

Taking into account the drift values given by GANZELMEIER et al. (1993) this may lead to safety distances up to 20 m (agriculture, horticulture) or 50 m (viticulture and hops) on the basis of the standard scenario (1 m width, 1 m length, 0.3 m depth = 300 l). In reality this scenario is rather seldom. Only some of the surface waters are fully and directly connected with arable land and in many cases a well developed bank vegetation exists. Besides many surface waters are flowing waters and the width and depth of the water body as well as the flow speed influence the pesticide concentrations in the real world significantly. Taking into account these realistic conditions the following conservative estimates can be made:

- A standing water, 30 m wide and 1 m deep, has 30,000 l water per m of length.
- A running water, 4 m wide, 0.75 m of deep, with 10 water renewals per day also to has 30,000 l water per m of length.

This means that realistic pesticide concentrations for these examples are 100 times lower than calculated with the standard scenario now used. If this type of environmental information is available in an easy to handle form it may be possible to include the real conditions in the field in a flexible risk mitigation concept, which allows to reduce the maximum distances where possible. This approach can be highly supported if the environmental information is available in digital form and can be analysed using GIS.

GIS is an specific collection of computer hardware, software, and geographic data. It is designed to efficiently capture, store, update, manipulate, analyse and display all forms of geographic information. In more simple words one could also say that GIS is a computer system, capable of storing and using data describing places on earth. Using this platform in combination with digital environmental information it is possible to establish a local risk evaluation.

Materials and methods

The tools necessary are

- a hardware platform with 600 MHZ processor, 256 MB RAM, and 20 GB hard disk storage
- ARC Info® together with supporting tools like ARC-View® or equivalent products, allowing for geo-referencing of data
- digital environmental information on
- land use: location and kind of agricultural crop
- digital elevation model taking into account the slope gradient near surface waters
- location of surface waters
- depth and width of surface waters
- type of sole substratum
- water quality such as pH and existence of defined organisms like daphnia or algae
- quality of the peripheral zone and bank vegetation.

In a study carried out during 1998 and 1999 information was collected concerning environmental data available in different parts of Germany and for the country as a whole which are necessary for a flexible risk mitigation concept on a local scale (PFEIFFER et al., 1999). To obtain information a questionnaire was designed asking about different types and availability of environmental information and was sent to 170 different institutions. Such as agricultural advise, agricultural research institutes, state ministries as well as institutes for geographical survey, and departments for water affairs and environmental protection. Furthermore for the area of the Rhine valley in Rhineland Palatinate digital environmental data were collected and introduced into GIS.

Results and discussion

Analysis of the questionnaire

The recoil of the questionnaire was between 14% and 100%. 58% of the departments for agricultural advise responded and the recoil from the institutes for environmental protection, geological and geographical survey, water affairs and environmental protection was between 64% and 94%.

The analysis of the answers gave the following information:

Table 1 Environmental information available for a realistic local risk mitigation

Type of information	Data source	Availability	Important for
Water body position	ATKIS[1]	Whole Germany	
Arable land position	ATKIS, ALK[2], InVeKoS[3]	Whole Germany	Drift
Bank vegetation	German surface water mapping	Will be fully available in 2 years	Runoff
Water width	ATKIS	Whole Germany	Erosion
Water depth	Different data bases	Fully available in single states of Germany	
Sole substratum	German surface water mapping	Will be fully available in 2 years	Interflow
Water quality	Different data bases	Fully available in single states of Germany	
Slope gradients	DHM[4]	Whole Germany	Runoff
Soil types near water bodies			Erosion Interflow

[1]: Official topographic and cartographic information system; [2]: Digital land data base; [3]: Integrated administration and control system;[4]: Digital height model

Focus on the Rhine valley in Rhineland Palatinate

The Rhine valley in Rhineland Palatinate is a 150.000 ha region between the Rhine and the Palatinate forest in south-western direction and the French border in the south. It is an area characterized by intensive agriculture with approximately 24.000 ha of vineyards and some thousands of ha of orchards and horticulture. Both standing surface waters and flowing waters are found in this area.

For this region the digital information on land use, width and depth of surface waters as well as on the kind of bank coppice was introduced into GIS and analysed. An important basis is ATKIS, which is available for the whole of Germany and which contains detailed information on land use.

Figure 1 Area of analysis in Rhineland Palatinate

From this analysis the information can be extracted that only a small portion of the arable land meets surface waters directly and that many surface water areas are wider than 1 m and have a well developed bank vegetation.

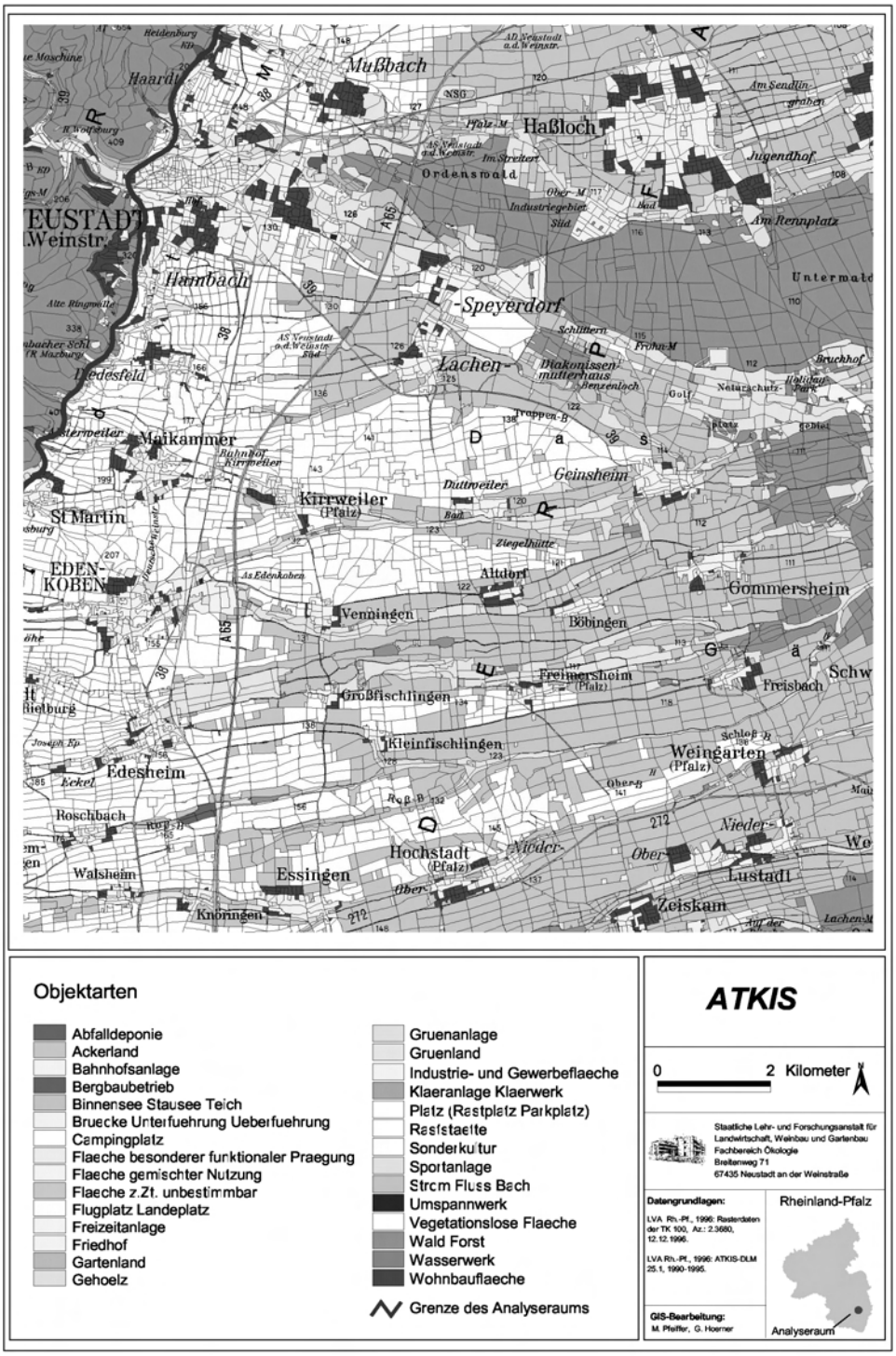

Figure 2 ATKIS data for the Neustadt region

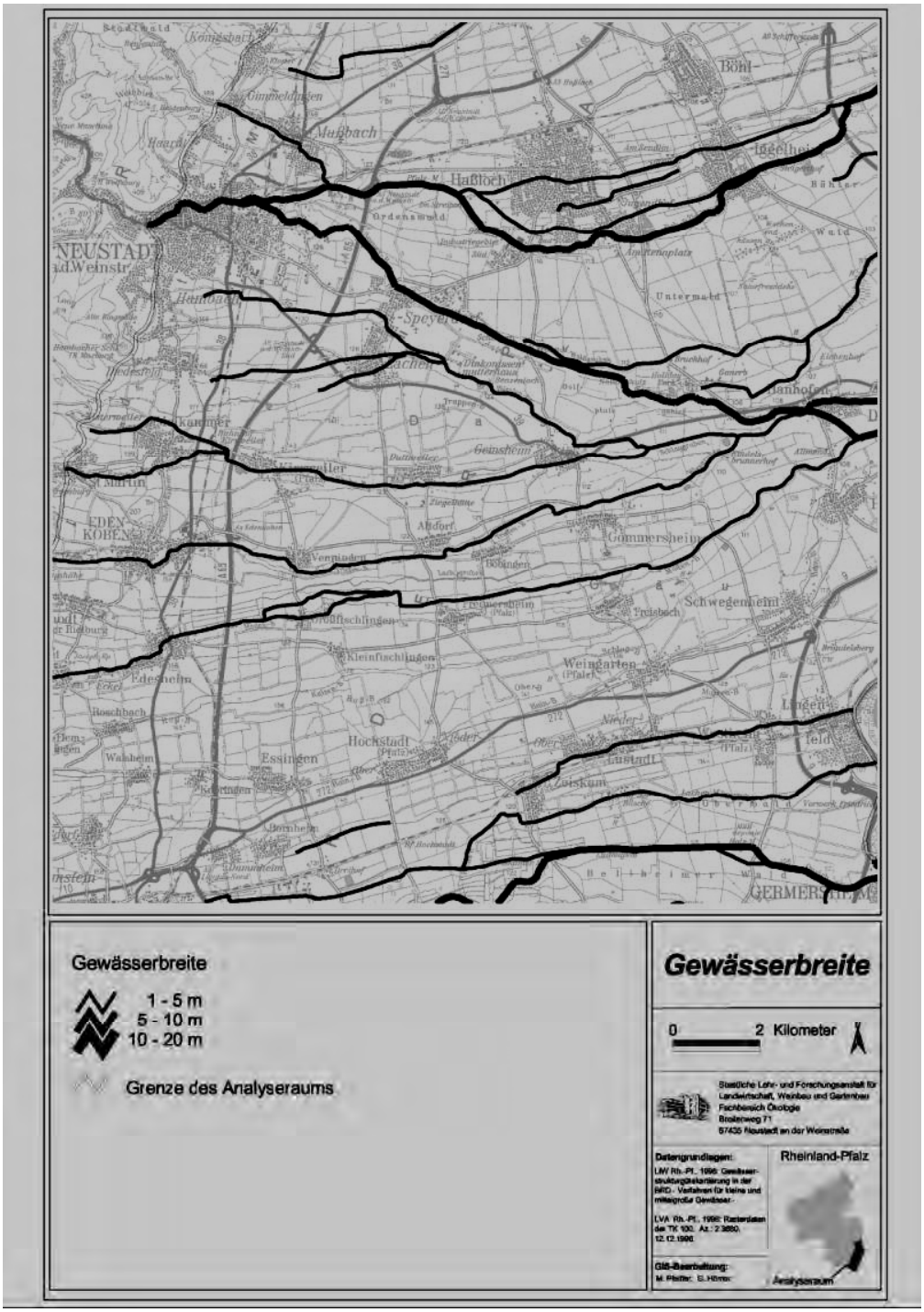

Figure 3 Width of surface waters in the Neustadt region

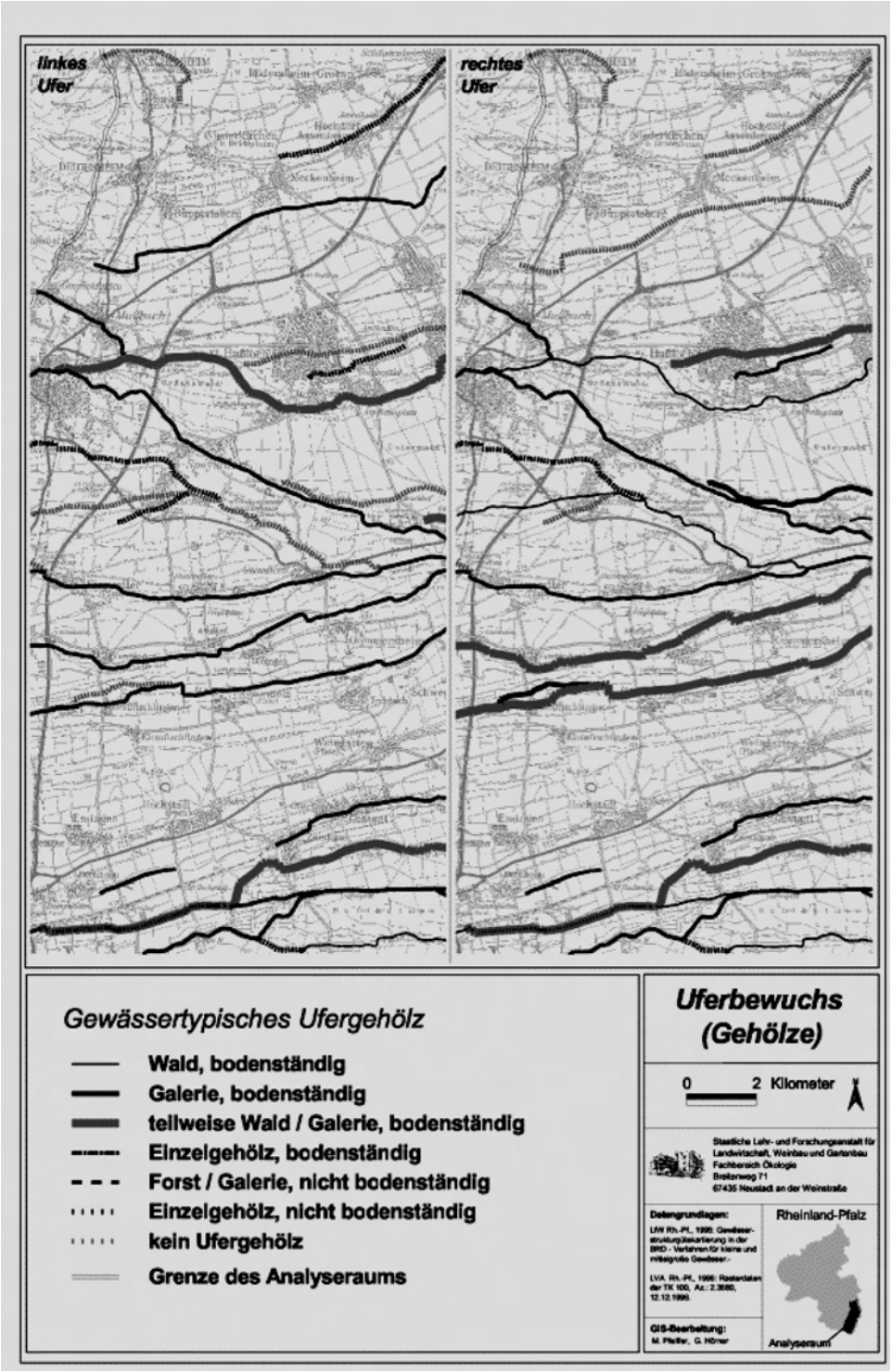

Figure 4 Bank coppice of surface waters in the Neustadt region

Local risk mitigation concept

The questionnaire and the local analysis of the Neustadt region may be used for a local risk mitigation following a detailed investigation of the environmental consequences of such an approach. Therefore a research project would be necessary to investigate the pesticide residues and the biology in surface waters in one or two test regions. The basic concept for such a project could be as follows:

Definition of environmental risk categories A, B, C, Z concerning the type of water body, its width and depth and the width and type of riparian vegetation. Category A would be a very safe region with a well developed riparian vegetation, and very wide and deep water bodies whereas category Z would fulfil the worst case estimates of the PEC determination as it is carried out today.

Environm. categories	Minimum distance in agriculture [m]	Minimum distance in special crops [m]
A	3	5
B	8	10
C	14	35
Z	20	50

Depending on the eco-chemical and eco-biological parameters (i.e. DT_{50} and C/TRC) of the pesticides a geographically related risk classification could be performed taking into account both the environmental conditions and the pesticide behaviour. To ensure that all requirements concerning law, practical handling, control and easy to understand characterisation on the package are met, the following scheme could be used:

Risk category	compound related parameters	possible environm. Categories	package print
I	DT_{50} or C/TRC low	A, B, C	green
II	DT_{50} or C/TRC medium	A, B, C, Z	yellow
III	DT_{50} or C/TRC high	B, C, Z	red

After the end of the research project the results would be open for discussion concerning the feasibility and the environmental consequences of the approach described.

Acknowledgements

The authors thank the BBA, Braunschweig for financial support.

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Risk mitigation measures to protect aquatic life: Dutch approach

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Introduction

The risk assessment of pesticides in The Netherlands is based on the Pesticide Act 1962. In February 1995 new regulation with respect to environmental aspects came into force. In this regulation criteria for persistence, leaching to groundwater and aquatic organisms are established.

In August 1998 new drift deposition figures for surface water and risk mitigation measures to protect aquatic life has been implemented in the regulation mentioned above.

With regard to terrestrial life much less risk mitigation measures are possible in the actual situation.

Aquatic Risk Assessment

The criteria for the risk assessment for aquatic organisms are mostly identical to the decision making criteria of the Uniform Principles. There is one major difference; for algae the NOEC-value is used instead of the EC50-value. The criteria are as follows:

- acute toxicity: PEC < 0.01 x L(E)C50 for Daphnia and fish and PEC < 0.1 x NOEC for algae;
- chronic toxicity: PEC < 0.1 x NOEC for Daphnia and fish.

The PEC calculation for surface water is at this moment only based on drift. Run-off and drainage are not taken into account, but there are activities going on to implement these routes of exposure in the exposure model for surface water (model is TOXSWA).

Drift Deposition Figures

There has been quite a lot of drift research in the last six years in The Netherlands. This research has been done by the IMAG (Institute of Agricultural and Environmental Engineering). As a result a rather large database with measured data on drift is available. Measurements have been done in apples, lawn trees, bulbs, potatoes, cereals, sugarbeets (row sprayer) and corn (row sprayer) (HUIJSMANS et al., 1997, 1999; VAN DER ZANDE, 1995; VAN DER ZANDE & HOLTERMAN, 1996; PORSKAMP et al., 1995). Within this research a comparison between different application techniques have been made.

Based on the research mentioned above a table with standard drift deposition figures for surface water has been implemented in the environmental regulation from February 1995. The figures are mentioned in Table 1.

Table 1 Standard drift deposition figures currently used in NL

application	Drift deposition on surface water in %
Orchards	17 before May 1 (no leaves) 7 from May 1 on (leaves)
Lawntrees	17 before May 1 (no leaves) 7 from May 1 on (leaves)
Field crops (incl. small fruit)	5
Horticulture (small)	1.6*
Bulb culture	0.2*
Green house	0.1
Aerial spraying	100

Packages of drift reduction measures are included in these figures. That's why these figures are rather low.

Risk Mitigation Measures

Because the calculation of the PEC for surface water is only based on drift deposition at this moment, risk mitigation relates to reduction of drift deposition on surface waters. With regard to drift reduction measures the following conditions are valid:

- the drift deposition must be preferably measured in the field or, if that is not possible be estimated in a reliable way, using a computer-model. The IMAG has developed such a computer-model, called IDEFIX, which is in the process of validation now;
- the drift deposition data must be relevant to Dutch agriculture and the equipment used;
- the drift reduction measures have to be reasonable in a way that enforcement must be possible in a normal way.

In The Netherlands the following drift reduction measures are fulfilling the above mentioned conditions. They relate to application techniques, buffer-zones and label restrictions.

Application techniques

- Orchards
  Tunnel sprayer:
  85% drift reduction;
  windbreak of trees:
  before May 1: 70% drift reduction;
  from May 1 on: 90% drift reduction.

- Field crops
  air assistance: 50% drift reduction;
  shielded bed sprayers: 99% drift reduction;
  drift reducing nozzles: 18% drift reduction;
  this figure depends very much on the type of nozzles;
  this figure is chosen at the lower end of the range edge nozzles:14% drift reduction.;
  row sprayers: 50% drift reduction;
  this is a temporary value, because more research has to be done with regard to this technique;
  exclusion of aerial spraying.

Buffer zones

In The Netherlands buffer zones till about 4 m are realistic, because of the relatively small pieces of arable land. The following conditions are valid for these buffer zones:

- Buffer zones are not grown with the crop which is grown in the center of the field;
- another crop is allowed on the buffer-zone as long as it is not sprayed by the plant protection product for which the buffer-zone is installed.

The amount of the drift reduction depends on the size of the buffer zone. In table 2 are the drift reduction percentages showed which belong to different sizes of buffer-zones.

Table 2 Buffer zones and drift reduction

Size of buffer zone in m	Reduction of drift deposition in %
1	52
2	67
6	ca. 90

Label restrictions

Also label restrictions will reduce the drift deposition. The following restrictions are possible:

- restrictions with respect to maximum application rate;
- restrictions with respect to maximum application frequency;
- restriction with respect to application time (e.g. orchards).

Within the near future (1/1/2000) generic rules with regard to drift reduction measures will be implemented which applies to every farmer. These rules have mostly to do with field crops and consist of packages of drift reduction measures, which are set up by arrangements between the government and farmer organisations.

With regard to these generic rules the following is relevant:

- 90% reduction of the drift deposition on surface waters is intended as a first step;
- the packages of drift reduction measures consist of a combination of techniques and buffer-zones (Figure);
- there are differences between groups of cultures;
- the farmer has a choice between (a limited number of) measures.

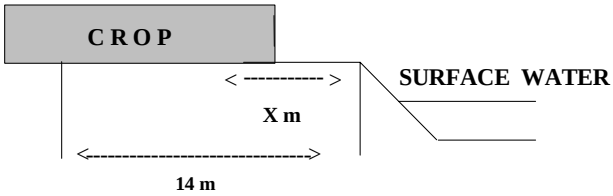

Figure Schematic picture of a buffer zone

The width of X depends on the type of crop and the use of drift deposition reducing techniques.

As an example the size of the buffer-zone in combination with the application technique for intensively sprayed crops is shown in Table 3.

Table 3 Size of buffer zone in combination with application technique for intensively sprayed crops

Crop/Culture	Size of X m	Technique
Intensively sprayed crops	2.25	none
	1.5	air assistance shielded bed sprayer
	1.0	plastic shields at the end of the field hand carried spraying boom

Within the buffer-zones only weeds may be controlled, but only by means of:

- knapsack sprayer with shielding;
- equipment which does not produce a spray.

Within the 14 m zone the following restrictions are valid:

- spray boom height < 50 cm;
- use of drift reducing nozzles *;
- use of edge nozzles *;
- wind-speed < 5 m/s * #;
- no application by means of an aircraft.

Notes: * this is not necessary in the case that a shielded bed sprayer is used; # this is not necessary in the case of severe danger of loosing the crop

The above-mentioned packages of drift reduction measures for field crops are already taken into account. Therefore the drift percentage for field crops is 1%.

Terrestrial risk assessment

Birds and mammals

At this moment there are not many possibilities for risk reduction. The following measures are feasible:

- lowering of the application rate;
- lowering of the frequency of application;
- a limitation of the time of application (e.g. only application in autumn in the case there is reproductive risk to birds);
- in the case of granules/seeds there are application methods which cover the granules/seeds. But there will always stay granules/seeds on the surface.

Bees

Apart from the reduction of the application rate and the frequency of application there can be restrictions on the label. In the Netherlands a ‘bee-sentence’ is put on the label in the case of high risk for bees: ”Do not apply to flowering crops or to crops when these are actively visited by bees. Do not apply when flowering weeds are present.

Other non-target arthropods

The following risk mitigation measures are possible:

- no use during specific periods of the year or the day when the organisms might be exposed;
- no use in specific crops or areas where the organisms might be exposed;
- no use within a certain distance from relevant areas (buffer zones);
- no use with application methods/equipment/formulations by which organisms might be exposed;
- no use over a maximum dose or frequency.

There are discussions going on with regard to the aim of the assessment for non-target arthropods: in-crop or off-crop assessment. In the case of an off-crop assessment drift reduction measures will be very important as a risk mitigation measure for non-target arthropods.

Earthworms and soil micro-organisms

At this moment there are not many possibilities for risk reduction. The following measures are feasible:

- lowering of the application rate;
- lowering of the frequency of application;
- in the case of glasshouses there can be a limitation to substrate cultures.

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Risk assessment and risk mitigation for non-target organisms in Austria

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Risk assessment

Risk assessments are conducted by utilizing laboratory acute and chronic toxicity data and comparing the results with predicted environmental concentrations (PEC) in surface water. Resulting toxicity exposure ratios (TERs) are then compared with the relevant Annex VI 91/414 EEC trigger values. If the trigger is breached, the risk assessment is refined and/or risk mitigation measures have to be taken in consideration.

Refined risk assessment

Consideration of fate and behaviour data like degradation or partitioning into sediment.

Testing of several different species can reduce the safety factors (down to 10 for acute toxicity) because the uncertainty of different susceptibility of different species is decreased.

Submission of data such as from microcosm or mesocosm can reduce the safety factor (between 10 and 1) depending on the quality of the study.

The reduction of the safety factor depends on expert judgement. Proposals for refined risk assessment are based on the guidance document ”HARAP” – Higher Tier Risk Assessment for Pesticides, from SETAC EUROPE/OECD/EC Workshop 1998).

Risk mitigation measures

Buffer zones

The safety distances to water bodies are established individually for each product based on toxicity and on the drift values published by GANZELMEIER et al 1995. Distances up to 10 m in field cultures and up to 20 m in top cultures have been set as restriction when granting authorizations. Buffer zones are statutory and are clearly stated on product labels.

The control of observance of the buffer zones is in the competence of the federal countries.

§ 12 (2) Pflanzenschutzmittelgesetz 1997 (Austrian pesticide law) provides the mutual recognition of registration granted by another Member State. So far Austria recognizes authorizations of plant protection products only from Germany. Restrictions of buffer zones in Germany (field cultures 20 m, top cultures up to 50 m) differ from those in Austria. This sometimes causes disagreement with national authorizations (§10).

An authorization according to §12 is granted by the ministry of agriculture and does not need the agreement of the ministry of environment. Problems not only arise concerning buffer zones but in general with the apply of Annex VI 91/414/EEC.

Mutual recognition of registration could cause problems especially during transitional periods and concerning subjects that are regulated at Member State level.

Regarding the problem of runoff no requirements (like grassed buffer zones or restricted allowance only in certain months) are made so far.

Application rate

The application rate is mainly determined by efficacy of the plant protection product against the pest.

For fungicides where multiple applications are usual the number of application is restricted by danger of persistence; in general the amount of application is less than in Germany because number of vines per ha is reduced (1000 l/ha instead of 1600 l/ha, no additional consideration of steep slopes).

For herbicides and insecticides the efficacy is most decisive. There mostly only one or two applications a year are made which can hardly be reduced.

Application technique

Spray drift can be reduced by recycling sprayers and new kind of nozzles (e.g., injector nozzles). Recycling units are used very seldom (about 10 pieces in Austria). In orchards in Styria the use is not possible because of the use of hail protection nets.

Information about spray drift reducing methods is brought to the farmers via the agricultural chambers.

Checking of the agricultural equipment (nozzles, pumps and manometer) is important but made on a more or less voluntary basis. It works very well in orchards, quite well in vineyards but less in arable crops.

In future there will be additional support for farmers working according to KIP- (Integrated Pest Management) and ÖPUL 2000- (Austrian Program to support an environmental fair, extensive and the natural living space protecting agriculture) Guidelines for checking their equipment periodically (every 3 years).

Local environment

Wind breaks, riparian vegetation, flow rate of water course and size of watercourse can influence the concentration of the plant protection product in the water.

The ecological ”quality” of the water-body could also be taken into consideration as well as the potential for recovery and recolonization.

These measurements for risk mitigation would in the responsibility of the farmer or local authorities and can hardly be ordered or controlled by the regulatory authorities.

The only possibility to have some influence is by advice through agricultural organization and would need the co-operation between several institutions. This is still a matter of the future.

Terrestric environment

Risk assessment for non-target arthropods

Legal basis

- Pflanzenschutzmittelgesetz 1997 (Austrian pesticide law)
- Council directive 91/414/EEC
- Commission directive 96/12/EC (amending Annex II, part A, 8. Ecotoxicological studies on the active substance and Annex III, part A, 10. Ecotoxicological studies)
- Council directive 97/57/EC (establishing Annex VI to directive 91/414/EEC)

The **risk assessment** is carried out according to EPPO Decision making scheme for the environmental risk assessment of plant protection products: chapter 9: arthropod natural enemies (1994) (tiered test system: 1. lab tests, 2. semi-field tests and (3.) field tests).

The **test species** and the **test procedures** should follow the SETAC Guidance document on regulatory testing procedures for pesticides with non-target arthropods (1994).

The **IOBC-classification** is used for the evaluation of the risk. In case of effects observed in the lab exceeding certain trigger values (lab tests: 30%) the notification has to establish through an appropriate risk assessment that under field conditions there is no unacceptable impact on those organisms after use of the plant protection products according to the proposed conditions of use. As recommended in the SETAC guidance document acceptability of effects should be assessed for ”within crop non-target arthropods” and ”off crop non-target arthropods”.

Based on the results of lab tests, semi-field or field tests there is an obligatory **labeling** of the plant protection product if certain trigger values are exceeded. This labeling is to inform the farmer. Two kinds of labels are used:

- ”the plant protection product is harmful to one or more certain tested species” (i.e. ”the plant protection product is harmful to the lady bird beetle”),
- ”the product is harmful to non-target arthropods”, this general labeling is obligatory if extensive testing has been done which resulted in a number of harmed species.

Risk mitigation measures

In addition to this kind of labeling special phrases are planned to manage ”unacceptable effects” for the in crop and off crop habitat. These measures to be taken are **restrictions** on the use pattern (for example: **in-crop habitat**: time or number of applications; **off-crop habitat**: appropriate application technique or buffer zone).

Since the implementation of the Council directive 91/414/EEC in national law no such labeling was prescribed so far because the notification although now obliged to fulfill this law is given a transitional period to deliver all necessary data.

Risk mitigation measures to protect aquatic life: German approach

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Introduction

The legal background for risk assessment on aquatic organisms in Germany is the Plant Protection Act from 1998 (ANONYMOUS, 1998) which refers to the EU-directive 91/414/EEC (EUROPEAN COMMISSION, 1991). Especially the annexes II, III and VI of this directive are important because the data requirement and the basic principle of the assessments are laid down in these documents (EUROPEAN COMMISSION, 1996, 1997). Furthermore, the risk assessment follows the proposals of the Draft Guidance Document on Aquatic Ecotoxicology in the frame of the Directive 91/414/EEC (8075/VI/97 rev 7). Concerning risk assessment there are some new developments especially with regard to refined assessment which should be discussed briefly.

Risk mitigation measures are the main topic of this paper. Buffer zones to protect aquatic life have been set as label restrictions in Germany for the last 10 years. Today nearly all plant protection products on the market in Germany are labeled with this type of restriction. These buffer zones have been set mainly on the base of one standard use situation which represented a reasonable worst case and was easy to handle in the authorization procedure. However, this approach has been overprotective in certain situations. As farmers may be charged with fines up to 100.000 DM if they don't follow these restrictions there is an urgent need to develop much more differentiated decision making schemes for setting buffer zones. A new scheme which has already been introduced into the authorization procedure in Germany will be described.

Toxicity assessment

The criteria for the toxicity assessment on aquatic organisms are identical to those of the Uniform Principles and the aforementioned Guidance Document. Data requirements are in line with annex II and III of the directive 91/414/EEC especially for the standard risk assessment. With respect to refined risk assessment ("unless" clause of annex VI) the recommendations of the SETAC-HARAP workshop are very important (CAMPBELL et al., 1999). After the workshop companies have submitted more frequently different types of higher tier tests which are in general more realistic than standard laboratory test systems. Microcosms - which means indoor systems except standard tests - are often used to conduct tests under more realistic exposure conditions. Only rarely are these used to determine recovery or to test a more realistic composition of life stages than the most sensitive ones which are usually tested in standard studies. Simple test systems containing sediment comparable with the ones used in spiked water test with sediment-dwellers have been very successful (STRELOKE & KOEPP, 1995). Acceptance problems have been identified in some cases with single species tests to show recovery in water-only-systems because indirect effects for example on predators are not covered. These methods should only be used if the TER-value is not too low, the recovery and recolonization potential of the organisms at risk are high and fast recovery can be shown. Cost efficient and flexible methods like the aforementioned should be used in situations where due to several conservative assumptions in the standard risk assessment the likelihood of effects is not too high. Microcosm tests usually don not lead to lower uncertainty factors because only one or a few species are tested. The main purpose is to generate more realistic toxicity values.

Probabilistic Risk Assessment (PRA) is another tool to be used in refined risk assessment (SOLOMON, 2000) but experiences so far in the authorization procedure are limited. A workshop (EUPRA – European workshop on Probabilistic Risk Assessments for plant protection products) to be held this year will bring more clarification in this area.

In general mesocosm tests (replicated outdoor systems containing a natural community, usually 2–10 m^3) are clearly the most relevant test method if substances are very toxic and do not fulfill the standard requirements of the Uniform Principles. Persistence and bio-accumulation are additional properties which complicate the risk assessment. The aforementioned test systems contain diverse assemblages of species. Therefore indirect effects can be determined. Furthermore recovery and recolonization is determined under realistic conditions.

Besides generating more realistic toxicity values mesocosm tests do lead to lower uncertainty factors because natural communities with a lot of species are investigated (CLASSIC-Workshop, HEGER et al., 2000). At the end of the toxicity assessment a decision upon the relevant toxicity value is to be made.

Exposure assessment

The exposure assessment is based in the first instance on the maximum application rate and the German drift values (GANZELMEIER et al., 1995) which were measured with conventional application technique. Recently new drift values have been published and these values are now used in the authorization procedure (RAUTMANN et al., 2000). Other exposure routes are considered on a case by case decision. With respect to the type of water-body a 30 cm deep stagnant system is regarded as representative. Drift reducing technique in tall growing crops has been considered separately especially in cases where an authorization was not possible for standard application technique. Recently a new official list of drift reducing techniques has been published (RAUTMANN, 2000) and is now used in the authorization procedure. At the end of the exposure assessment the Predicted Environmental Concentration (PEC) is determined.

Risk assessment

For the risk assessment the relevant toxicity value is divided by the PEC. This Toxicity/Exposure value (TER) is compared with the relevant trigger value of the Uniform Principles. If a refined risk assessment has been conducted the standard triggers are no longer decisive and a decision upon the most appropriate uncertainty factor has to be made.

Risk mitigation measures

As mentioned before risk mitigation measures like buffer zones to protect aquatic life have been set in Germany for the last 10 years and today nearly all products have such a restriction on the label. In general distances of buffer zones are up to 20 m for field crops and 50 m for tall growing crops. On the basis of the aforementioned new basic drift values distances of up to 250 m are possible. These large distances are mainly important for regions with a low density of water bodies where the use of products should be possible even if there is a high toxicity for aquatic organisms but exposure is not likely.

In principle the following conditions have been used as standard use situation for the setting of buffer zones:

- Maximum application rate.
- Standard application technique.
- Small stagnant 30 cm deep water body.
- Whole population is contaminated at the same time.
- All communities are of equal sensitivity.

These assumptions and especially the combination of all these conditions at the same time in the same water body are not very likely. Therefore discussions have been underway during the last 2 years to improve the procedure of setting buffer zones. A special group comprised of experts experienced in decision making in the authorization procedure and experts from the extension service who are familiar with the practical situation of farmers was formed to create a system which should be practically feasible. Furthermore the proposals should not only be scientifically sound but at the same time easy to handle in the authorization procedure, fulfill legal requirements and are enforcible (STRELOKE & ROTHERT, 1999).

The basic idea of the new scheme is to use not only one standard use situation as base for setting of buffer zones but to consider additional use situations which may lead to another risk for aquatic organisms than the standard one. The LERAP (Local Environmental Risk Assessment for Pesticides) system in the UK but also procedures established in the Netherlands or Sweden are in general comparable (NORMAN, 2000; VAN VLIET, 2000; ARVIDSSON & LJUNGSTRÖM, 1998). Like the LERAP approach those local conditions should be taken into account which are important for the degree of risk. There are quite a lot of conditions which may change the risk not only qualitatively but also quantitatively:

- The actual application rate used.
- The type of application technique.

- The type of water-body (lentic - lotic, small - large) with respect to dilution.
- The vegetation at the embankment.
- Bio-availability of the compounds.
- Recovery and recolonization potential in the water-body.
- Sensitivity of the effected community.

It is obvious that all these properties could not be written on the label. Furthermore one cannot expect farmers to follow such complicated risk mitigation measures. But this understanding by farmers would not only be important for the acceptance of the measures amongst them but also necessary from a legal point of view. Additionally it is expected that there would be a continuous development of the state of the art for all these issues but the restrictions should not be changed continuously to prevent confusion. Finally, preliminary calculations on the base of rough estimates for different use situations made clear that not all conditions and combinations of these conditions would lead to considerably different distances.

Considering all these arguments it was decided to define 4 different risk categories with a fixed degree of risk reduction compared to the standard situation. These categories A, B, C, D represent a risk reduction of 99, 90, 75 and 50%. The new buffer zones include not only a distance for the standard use situation but also for the 4 risk categories (Table 1). An official list of risk mitigating use conditions was established where the risk categories, the degree of risk reduction and use situations evaluated by the authorities responsible for the authorization procedure were connected in a legally binding way.

Table 1 Risk categories, degree of risk mitigation and typical use conditions as outlined in the official list of risk mitigating use conditions (Note: The official list includes only the first, the fourth and the fifth column!)

Risk-category	Risk-mitigation	Factor TER	Risk- points	Local use conditions
A	99%	100	20	No entry up to now
B	90%	10	10	Application technique with 90 % drift reduction
C	75%	4	6	Application technique with 75 % drift reduction Lotic water-bodies with a minimum width of 2 m
D	50%	2	3	Application technique with 50 % drift reduction Riparian vegetation with a minimum width of 1 m

In practice the TER values for the standard use situation are calculated first. Subsequently these numbers are to be multiplied with the factors of 100, 10, 4 and 2 to calculate the TER values for the single risk category. The distance where this value exceeds the relevant trigger value is set within the restriction as relevant for the special risk category (Table 2). A restriction is not set, if the TER values in a distance of 1 m in field crops and 3 m in tall growing crops is higher than the relevant trigger values of 10 or 100.

Table 2 Example of a use in orchards (application rate 0.5 kg/ha, early stage, new drift values, most sensitive organism Daphnia, NOEC of 10 µg/l, relevant trigger values 10)

Distance in m	Standard TER	Category A, Factor 100, TER	Category B, Factor 10, TER	Category C, Factor 4, TER	Category D, Factor 2, TER
3	0.2	20	2	0.8	0.4
5	0.3	-	3	1.2	0.6
15	1.1	-	11	4	2
30	6.0	-	-	24	12

Restriction on the label

Between the treated area and a surface water – except those which contain water only occasionally but including those containing water periodically – there must be kept the distance specified below when the plant protection product is applied. If the conditions defined in the official list of risk mitigating use conditions from April 27th, 2000 (FEDERAL GAZETTE from May 26th, 2000, page 9878) in the effective version are fulfilled then only the reduced distances are sufficient. For those categories marked with an asterik Article 6 (2) sentence 2 of the Plant Protection Act must be followed:

- Orchard: 40 m Reduced distances A - * m, B – 10 m, C – 30 m, D – 30 m".

If more than one condition is relevant for a single water-body the factors for the relevant single conditions are to be multiplied. To ease these calculations the factors were converted to logarithm and multiplied by 10. These calculated risk points (see Table 1) are simply to be added by farmers in order to identify the relevant risk category (Table 3). Example:

- Combination of application technique with 75 % drift reduction and running water-body: 6 + 6 = 12.
- Consequently the local situation belongs to the risk category B. Therefore the distance stated on the label for category B must be obeyed.

Table 3 Table to be used by farmers to find the relevant risk category for his special use situation especially in cases where more than one condition included in the official list is relevant for a single application

Risk category	Minimum number of risk points
A	20
B	10
C	6
D	3

As mentioned above the official list on risk mitigation use conditions currently contains the application technique, the type of water-body and the vegetation on the embankment as relevant conditions for differentiated risk mitigation measures. Especially on the effect side further conditions should be regarded as important in order to come to more reliable and realistic predictions of risk. However, currently there is no consensus whether sufficient scientific background exists to include more types of water-bodies in the official list and especially in higher risk categories. With respect to conditions like the potential of recovery and recolonization in a water body which are very difficult to identify by farmers additional information systems must be made available to farmers. Consequently research projects are underway or planned to check the practicability of the aforementioned new concept, to develop further scenarios and to establish Geographic Information Systems (GIS) in order to generate easy-to-understand information on the water-bodies for farmers.

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Aquatic environment

Herbicides in the catchment area of the haltern reservoir - monitorings and results

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Haltern waterworks and catchment area

The haltern waterwork, with an annual output of about 110 million cubic metres , provides the main part of the drinking water supply for around one million people, and for business and industry located on the border of the Münsterland. At the Haltern site, water-bearing sands located at depths down to 200 m possess optimimal geological conditions for natural water collection.

25% of the total annual output consists of uncontaminated groundwater. The largest part however comes from the surface water by artificial groundwater enrichment. The Stever river is the eastern influx to the Haltern reservoir. The soil in this area contains heavy clay and the Stever water is more contaminated with pesticides than the Mühlenbach river, the western tributary with more sandy soil (Fig. 1)

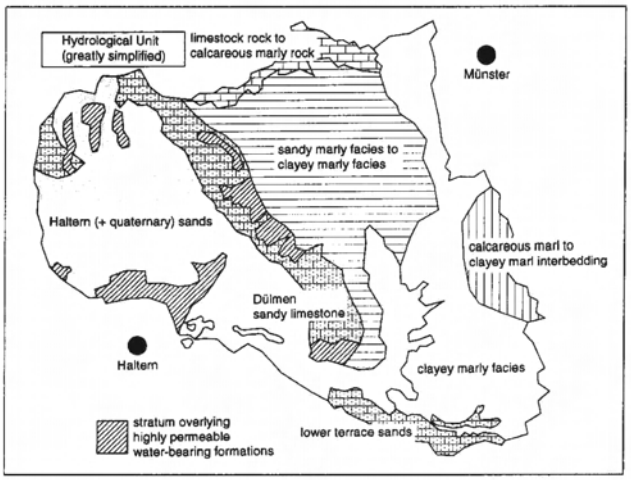

Figure 1 Hydrogeological conditions in the catchment area of the haltern reservoir

The Stever region is intensively used by agriculture, especially for the cultivation of grain and maize (Tab. 1).

Table 1 Agriculture in the Stever region

Total area		**880 km^2**	
Area used by		515 km^2	
Area under cultivation	(%)	426 km^2 (82.8 %)	
	% grain		192 km^2 (45.1 %)
	% maize		149 km^2 (35 %)
Pastures or meadows	(%)	89 km^2 (17.2 %)	
No. of farms		3000	

Monitoring studies

The application of herbicides to grain fields in spring and autumn or to maize fields in summer results in high concentrations of herbicide substances at some locations, especially in the Stever river. Routine analysis for about 160 herbicides and degradation products revealed that the contamination is confined to a relatively small number of substances.

Analyses performed regularly at the location where the Stever river feeds directly into the Haltern reservoir were targeted to several other substances in addition to triazines, urea herbicides, Bentazone and polar herbicides (e.g. phenoxyalkanoic carbonic acids) (Tab. 2).

Table 2 Herbicides in the Stever river in 1998

Herbicide	No. of Measurements	Arithmetic mean alues	Maximum values	Time of max. values
Isoproturon, µg/l	165	0.19	1.70	April
Chlorotoluron, µg/l	165	0.05	0.83	January
Methabenzthiazuron µg/l	165	<0.03	0.07	June
Atrazine, µg/l	166	0.03	0.27	June
Simazine, µg/l	166	<0.03	0.12	June
Propazine, µg/l	166	<0.03	0.05	April
Terbutylazine, µg/l	166	0.12	3.40	June
2,4-D, µg/l	34	0.03	0.11	June
2,4-DP, µg/l	40	0.03	0.19	May
MCPA, µg/l	34	0.03	0.22	June
MCPP, µg/l	40	0.03	0.10	April
Fluroxypyr, µg/l	36	0.03	0.10	June
Bromoxynil, µg/l	34	<0.03	0.11	June
Bentazone, µg/l	40	0.15	1.90	June
Metolachlor, µg/l	7	0.05	0,11	July
Dimefuron, µg/l	165	<0.03	0.03	October
Diuron, µg/l	165	0.08	0.64	June

In 1998 only 6 herbicidal substances were frequently measured at concentrations above 0.1 µg/l (Table 3).

Table 3 Occurrence of relevant herbicidal substances in the Stever region in concentrations higher than 0.1 µg/L

Herbicide	Crop
Isoproturon	grain
Chlorotoluron	grain
Bentazone	grain, maize
Terbutylazine	maize
Atrazine	maize

Furthermore, sampling stations were set up throughout the whole catchment area. In order to obtain a better overview about possible areas of high contamination in the region, samples were taken from these stations and analyzed at least once a month. The selection of sampling stations was made according to the following criteria:

- contamination of the body of water;
- knowledge of particular contamination routes;
- location of possible point sources.

Water samples were generally taken randomly. Automatic sampling devices were employed wherever their use was dictated by special programs or situations. Contamination maps were drawn on the basis of the arithmetic mean values (Fig. 2).

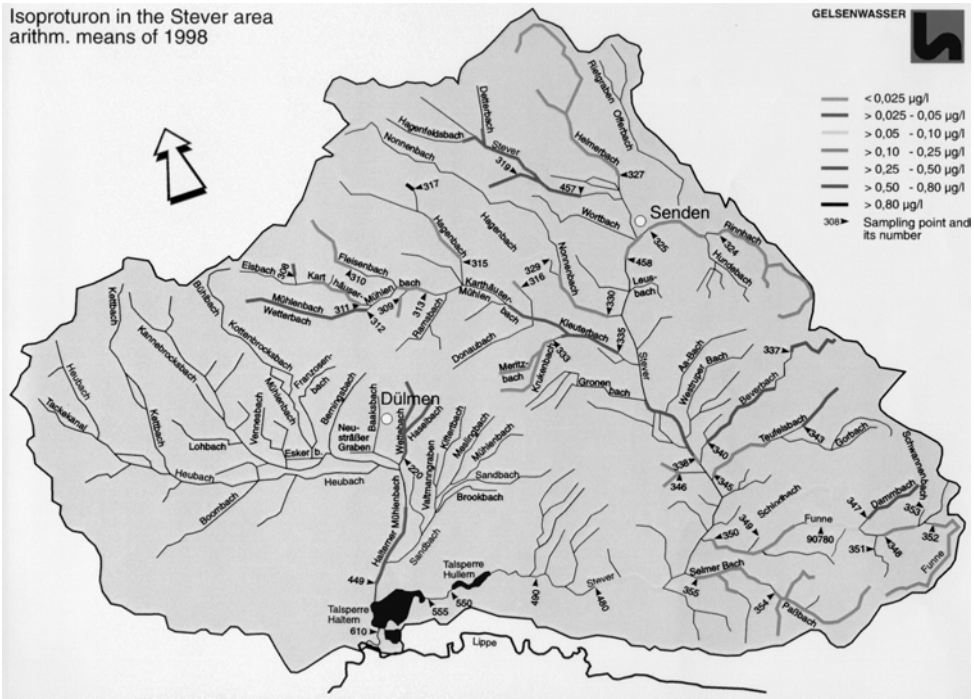

Figure 2 Contamination map of Isoproturon

Various informations, e.g. areas of high contamination and the impact of point sources, can be easily read from these maps, which also serve as the basis for elaborating specific minimization strategies. Such a strategy is an important part of the agricultural advisory services.

Special programs for minimizing herbicidal contamination

Although the overall advisory strategy for reducing the input of pesticides in the surface water by advertisement and the demonstration of alternative ways for application enjoyed a modest success, it did not bring about dramatic change. It was concluded very early that the only way to sustained minimization was to find substitutes for those herbicidal substances which tend to leach out easily and which contribute the most to the contamination. It became evident that herbicidal concentrations in water could be reduced only by finding feasible substitutes for such herbicides which pose a problem because of their unfavorable physical and chemical properties and their large overall use.

Substitution of urea herbicides

The first project for substitution was carried out in cooperation with herbicide producers, within a defined subcatchment area, the Karthäuser Mühlenbach (Fig. 3). The aim was to replace urea herbicides used in grain cultivation, such as Chlorotoluron and Isoproturon, with alternative substances exhibiting a lower tendency to leach or wash out. The project duration was limited to the period from autumn 1994 to spring 1997.

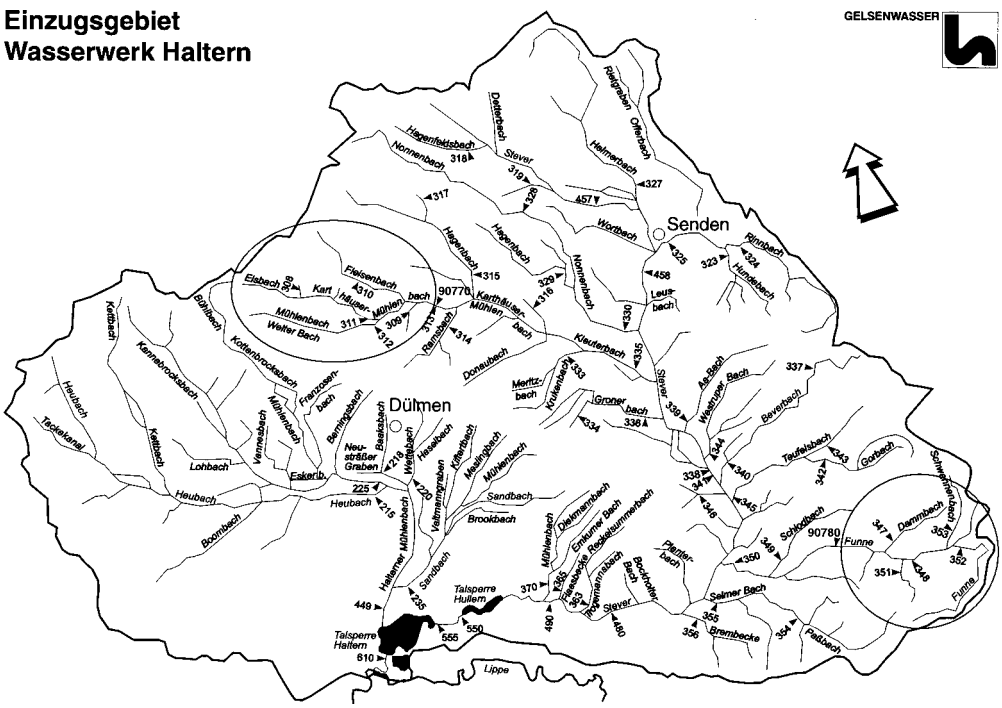

Figure 3 Location of the Karthäuser Mühlenbach (left) and the Funne (right)

As a comparison to the situation, where no substitution of herbicides had taken place, measurements were performed in another subcatchment area, that had previously exhibited practically the same degree of herbicidal contamination (Funne region).

The results were documented by measurements. Automatic sampling devices were used to achieve time-controlled sampling over a 24-hour period.

In addition, the water samples taken during the study were analyzed to ascertain whether they contained the substitute herbicides and to exclude further contamination.

The cooperation was expecting, that the farmers in the Funne region, which was serving as a comparison area, would also use those herbicide substitutes, as soon as they would learn about the agricultural and financial advantages of the use of the substitutes even without the help of the cooperation. However, it showed, that this minimization effect was not pronounced in the Funne region as compared to the central project area of the Karthäuser Mühlenbach.

It could be shown, that the concentrations of Isoproturon and Chlorotoluron decreased significant by this experiment. The acceptance by the farmers was very good. In the Karthäuser Mühlenbach area the sunstitution product was applied on nearly 88 % of the area where grain was cultivated

On the basis of the distinct success achieved, the study was subsequently extended to include the former comparison area (Funne region). However, this part of the study was carried out under improved conditions because additional substitute herbicides had become available at that time.

It is planned to further extend this minimization strategy to cover the whole Stever region, starting in the autumn of 1999. Because of the size of the area and the number of farms, this will certainly not be an easy task (Table 4).

Table 4 Large-scale study on the replacement of urea herbicides

Region	Period of time	Area (ha)[1]	Substitute herbicides
Karthäuser Mühlenbach	9/94 to 6/97	4260	Ralon, Ralon Super
Funne	9/97 to 6/99	3653	Ralon, Herold, Cadou, Topik, Stomp, Bacara, Lexus class
Stever	9/99 to ?	88000	Tolkan flo, Stomp, Boxer, Igran, Treflan, Fenikan, Herold, Bacara, Lexus class, Topik, Ralon Super

[1] ha = hectare = 2.47 acres

The study results obtained so far show that:

- A sustained minimization of urea herbicides can be achieved only by replacing these substances by other herbicides.
- So far the alternative herbicides have not been detected in the water.
- The farmers have not experienced any financial disadvantages or disadvantages with respect to agricultural technology. Instead the substitute herbicides have ultimately proven to be cheaper and more effective.
- The farmers displayed a high acceptance of these alternative herbicides as long as financial assistance was given by the project.

Minimization of Bentazone contamination

The application of a new herbicide to maize crops in 1998 resulted in massive contaminations of the Stever river with Terbutylazine and Bentazone (Fig. 4).

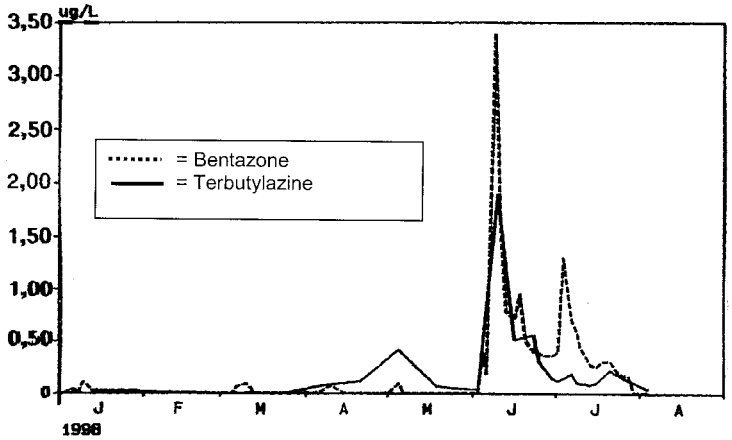

Figure 4 Terbutylazine- and Bentazone- concentrations in the Stever river in 1998

The joint strategy elaborated by the cooperative and the producer envisaged that farmers and other herbicide users should refrain from applying this agent in 1999. This strategy was incorporated in the recommendations of the Chamber of Agriculture and the producer.

The analysis data for 1999 revealed a notable decline in the concentrations of these two herbicides. Despite this impressive success, the following questions remain:

- Is the decline due to the fact that the majority of farmers followed the recommendations? or
- Is it due to the meteorological conditions prevailing in 1999?

Summary

By means of selective monitoring in the Stever Region, it is possible to identify areas of high contamination and to characterize the type of contamination (e.g. point sources) more precisely. This monitoring is the basis for selective minimization strategies and for advisory services to agriculture.

The first trials performed with substitute herbicides have brought about excellent results. Moreover, these trials have shown clearly that the physical and chemical properties of a herbicidal substance are decisive in determining the probability of a contamination. Consequently, only substances meeting the corresponding requirements may be used in certain high sensitive areas, e.g. areas characterized by steep inclines and heavy soil.

These observations must be taken reasonably into account in the approval procedures. The logical conclusion would be that such approval should only be granted for only a limited period of time. Should contamination then be detected again in certain calibration areas, application of the substances in question shoud be restricted.

Effects of the use of plant protection products on aquatic species – a monitoring approach

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Abstract

In order to obtain a more realistic picture of the real effects of agricultural practice and crop protection use on macroinvertebrate communities of neighbouring aquatic systems, a retrospective approach similar to the RIVPACSystem was chosen: In two agricultural regions in Germany differing in crops and cultures, ditches and small brooks were sampled for the benthic macroinvertebrate communities. Samples were collected at 40 sites of each region, differing in the predictable loads from the use of crop protection products in the drainage area. Based on these data, the sites were clustered by using the TWINSPAN approach. By multiple discriminance analyses (MDA) of the site clusters including habitat and exposure parameters, the main factors influencing species composition in autumn 1998, in spring, summer and autumn 1999, and in spring 2000 have been and will be identified. The results of the applied statistical approach are compared with results of a principle component analysis (PCA) followed by the principle response curve approach (PRC).

First results from the region ”Altes Land” near Hamburg, which is characterized by extremely high exposure of surface waters and a very definite exposure situation, were obtained from the autumn 1998 and spring 1999 samplings. No significant effect of land use patterns or exposure on species richness could be observed. With respect to species composition, significant effects could be observed in autumn 1998 and spring 1999. They were less significant than those by a few other influence factors (e.g. water depth), and decreased from autumn to spring. The results of the two statistical approaches have to be compared with respect to the single influencing factors. The taxa being sensitive towards the different influences will be identified. The inclusion of the summer data will clarify the extent of the acute effects, the inclusion of the next autumn and spring data will enable the quantification of the recovery. The data set of the second region will be evaluated in the same way to enable comparisons and clarify possibilities of extrapolation.

Introduction

The predictive risk assessment for plant protection products is based on realistic worst case assumptions concerning potential exposure and effects without including the probability of occurrence. The aim of the project is to get an impression about the realism of predictive risk assessments by using a monitoring approach which is based on the RIVPACS methodologies (MOSS et al., 1987; WRIGHT et al., 1993; WRIGHT, 1995): At first, the benthic macroinvertebrate communities are investigated at each site. Based on the community data, the sites are grouped, and relevant differences between communities at different groups of sites are detected. In a second step, hints to the main factors causing these differences are obtained by relating the result of the grouping to site specific values of defined habitat parameters. The main questions to be answered are: Does the use of plant protection products have a significant impact in the long run on communities in small surface waters of intensively used agricultural regions? Is it more or less important than the impact by other influence factors?

Two agricultural regions were chosen following different objectives. The region ”Altes Land” near Hamburg is a completely anthropogenic landscape with very intensive agriculture by orchards with high spray drift rates of plant protection products. A net of drainage and irrigation ditches of very high density leads to an extremely high exposure of surface waters calculated to contribute about half of the total masses in Germany (BACH et al., 1999). Thus, the region represents a worst case exposure. On the other hand, the very uniform ditches, the long and narrow pieces of land separated by them and the permanent cultures with few sorts result in a very clear exposure situation, representing the ”best case" concerning exposure calculations as well as statistical evaluation and interpretation of effects. They are well suited for the validation of applied methodologies concerning site choice, parameter choice and statistical approach.

The ”Region of Braunschweig” was chosen as a representative region for intensive use of field cultures in a landscape dominated by agriculture and structured by drainage ditches and regulated brooks and thus for a

typical case of intensive agricultural use. The main objective of the study part performed in this region is to investigate the possibilities of extrapolation from the methodology and results of the ”Altes Land” region to regions with clearly more heterogenic land use patterns and water net structures.

Methods

Choice of sites

The sites should:

- (partly) be neighboured by cultures which ensure exposure to plant protection products with direct effects on the investigated community (insecticides, fungicides),
- show clear differences of the individual sites with respect to predictable loads caused by the use of plant protection products,
- be as homogeneous as possible concerning natural and further anthropogenic influence factors being important for the benthic macroinvertebrate community structure,
- be permanent aquiferous,
- guarantee a sufficient base for multivariate statistics: 40 sites per region, each located at an individual water body.

The 40 sites selected in the region of ”Altes Land” were:

- five blind ditches within old apple cultures, which are no longer used for apple production (two of them with one side directed to other cultures),
- nine blind ditches within controlled biological apple cultures
- (two of them with one side directed to other cultures),
- 13 blind ditches within integrated apple cultures,
- 13 blind ditches within greenland.
- all blind ditches are draining into collecting ditches.

The chosen sites in the ”Region of Braunschweig” are mainly neighboured by cereal and sugar beet cultures or greenland. 32 of the sites are located in sand and loess regions around Braunschweig, eight sites are located in the Weserbergland near Hameln.

Sampling and determination of the macroinvertebrate community

The sampling dates were and will be: autumn 1998 (after all treatments), spring 1999 (before the first treatment), summer 1999 (just after main insecticide treatment), autumn 1999 and spring 2000 (for comparisons between years). The sampling is performed by using nets of 1 mm mesh width and conducted separately for sediment and macrophyte habitats at each site. In Braunschweig, surber samplers are used. The taxonomic resolution is the species level, where possible, and oriented at the guideline used for the determination of the Saprobienindex.

Investigated habitat and exposure parameters

The measured habitat parameters at each site are temperature, oxygen content, pH, conductivity, hardness, alcalinity, calcium, magnesium, copper, iron and manganese contents, DOC, nitrate, nitrite, ammonium, total nitrogen, total phosophorus, o-phosphate (all measured two (”Altes Land”) or three (Braunschweig) times before biological sampling); TOC, structure and depth of the sediment, stream velocity, water depth, degree of shading/woody plants on the bank, degree of macrophyte coverage, dominant macrophyte species, last cleaning, distance to the next collecting ditch (only in ”Altes Land” region), direction (only in ”Altes Land” region).

The exposure was calculated based on the type of and distance to the neighbouring cultures, the vegetation heigth of the riparian zone, the degree of coverage of the water surface by macrophytes, water width, water depth, and runoff potential (only Braunschweig).

Statistics

The influence of individual parameters on species richness was tested by forming classes of values of the parameter looked at and calculating the statistical significance of differences of species numbers at sites measured with parameter values belonging to different classes by using the Tukey test (BOX et al., 1978).

Based on the taxonomic data (species level where possible), the sites were clustered by using the TWINSPAN approach (dichotomic clusters; based on correspondence analysis (HILL, 1994). By multiple discriminance analyses (MDA, SPSS programme package) of the site clusters including habitat and exposure parameters, the main factors influencing species composition have been and will be identified. The results of the applied statistical approach are compared with results of a principle component analysis (PCA) followed by the principle response curve approach (PRC, VAN DEN BRINK & TER BRAAK, 1999). The canonical coefficients describing species composition are related to site classes with distinct intervals of values of individual parameters. One class of values (either the highest or the lowest ones) is regarded as reference and set 0, whereas the others are expressed by the differences to the reference. This kind of statistical evaluation is performed with the taxonomic data after having been aggregated to family level. Statistical significance was tested by applying Jackknife, a Monte-Carlo Method (EFRON & GONG, 1983).

Thus, it should be possible to evaluate the influence of specific anthropogenic impacts in comparison to other influences, either anthropogenic or natural ones. The presented approach is based on the statistical analysis of unweighted data and therefore different from indication concepts in monitoring programmes. It should be able to detect and describe effects on species richness and composition of aquatic macroinvertebrates.

Preliminary results and Discussion

The presented results are preliminary results of the running study. At this stage, only the two first sampling periods of the ”Altes Land” region have been roughly evaluated statistically.

Species richness

In the region ”Altes Land”, the clearly differentiated taxa found during the sampling in autumn 1998 were 133 species, genera and families. In spring 1999, the respective number was 99. The dominating taxonomic groups were *Coleoptera*, *Oligochaeta*, and *Gastropoda*.

No significant effect of land-use patterns or exposure on species richness was observed in autumn 1998 and spring 1999.

Species composition

The discriminance analyses (MDA) of the first separation level of TWINSPAN show a main influence of distinct parameters on species composition, which together form the discriminance function by which the correct prediction of site classification by TWINSPAN is maximized (Table 1).

Some parameters are physico-chemical ones, others are related to the recovery potential (state of recovery) of the community at the individual site (distance to the collecting ditch, last cleaning), some parameters are related to the degree of shading or the accessability for insects (degree of shading, greenland use/distance to orchards). The parameter exposure by integrated culture, which was only calculated for the MDA, was found to be one of the important parameters for the prediction in spring. It integrates water width and depth, distance from integrated cultures and other factors diminishing exposure. In autumn 1998, water depth was sufficient to explain the influence of exposure, and/or other properties of water depth for the community structure were more important.

The principle component analysis followed by a principle response curve analysis over time (PRC) resulted in pointing out significant influences of only five parameters (Table 2). These can be distinguished into to different groups: One with parameters showing continuous value-response relationships (Figures 1 and 2) and the other with parameters showing a specific window of values being significantly different from the reference class (highest or lowest values).

Table 1 Main influence parameters, ranked according to the absolute values of their standardized canonical discriminance function coefficients. *Italic*: influence correlated with recovery potential. Dark shading: direct influence of exposure. Medium shading: influences potentially correlated with exposure. Light shading: Influences correlated with irradiation and accessability for insects. Not marked: physicochemical parameters

	Autumn 1998	**Spring 1999**
First TWINSPAN separation level Correct prediction of sites by discriminance function	1. Distance to the collecting ditch	1. Last cleaning
	2. Calcium concentration	*2. Distance to the collecting ditch*
	3. Oxygen concentration in water	3. Oxygen concentration in water
	4. Water depth	**4. Exposure by integrated culture***
	87.5 %	90 %
	(85 % at cross validation)	(87.5 % at cross validation)
Second TWINSPAN separation level Correct prediction of sites by discriminance function	1. pH value	1. Ammonia value
	2. Conductivity	2. Oxidative/reduced sediment
	3. Degree of shading	**3. Greenland use**
	65 % (55 % at cross validation)	75 % (70 % at cross validation)

* additionally calculated only for the multiple discriminance analysis

Table 2 Significant influence parameters revealed by principle response analysis over time (autumn 1998 – spring 1999). *Italic*: influence correlated with recovery potential (in brackets because there is no significance if the only actually cleaned site is left out). Dark shading: direct influence of exposure. Medium shading: influences potentially correlated with exposure. Light shading: Influences correlated with irradiation and accessability via air. Not marked: physicochemical parameters

Continuous value-response relationship	**Specific window of values with maximal response**
Water depth	(Last cleaning)
Type of culture (old-biological-integra ed)	Distance to orchards (greenland use)
	General exposure
	Degree of shading

The striking differences between (-) and similarities of (•) both statistical evaluations are:

- The complete lacking of physicochemical parameters as significant influences identified by PRC. This might be due to the uniformity of the ditches resulting in low differences and due to the different handling of distributions and mean values by both methods.
- The fact that the distance to the next collecting ditch is the most important influencing factor identified by MDA, but not a significant influence by using PRC.
- Water depth is identified as one of the most important influences by both approaches.
- The exposure by integrated cultures and some parameters correlated with exposure are identified by MDA as influencing factors; integrated culture and general exposure by PRC.
- Greenland/shading are identified as minor but significant influences by both approaches.

The factor ”last cleaning” is identified to be important by both approaches. However, the importance is triggered by one site which was cleaned between the two sampling periods. If this site is left out in the spring data set, there is no significance left concerning the time intervals between 1-4, 4-7 and above 7 years. This can be used as further indication for the high recovery potential of the regional community.

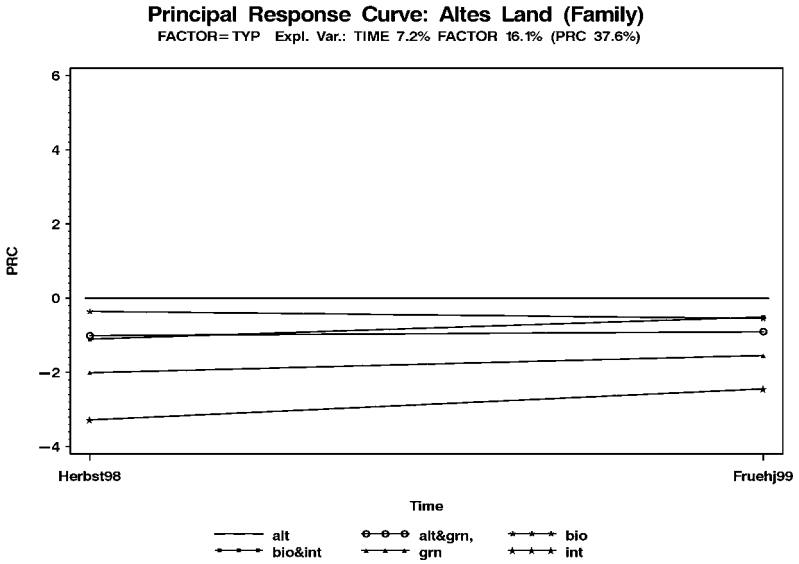

Figure 1 Principle response curve figure of culture type influencing the aquatic community (family level). Alt: old apple cultures no longer used for apple production; bio: controlled biological apple cultures; int: integrated apple cultures; grn: greenland. The sites within integrated cultures are significantly different from old and controlled biological cultures

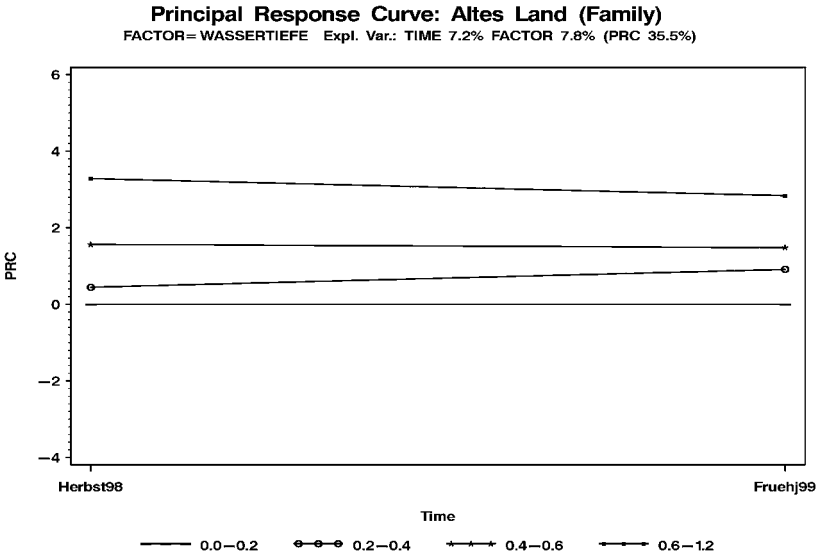

Figure 2 Principle response curve figure of water depth (m) influencing the aquatic community (family level). Deep sites are significantly different from shallow ones.

Interim conclusions

In autumn 1998 and spring 1999, no significant effect of integrated plant protection on species richness of the community of aquatic macroinvertebrates could be observed.

Significant effects on species composition:

- could be observed in autumn and spring,
- were not more pronounced than those by a few other influence factors,
- decreased from autumn to spring.

First results of the summer samples demonstrate clear effects on arthropods at the sites surrounded by integrated cultures. Thus, a great recovery potential can be postulated, which has to be confirmed or rejected by the next two data sets.

Outlook

The results of the two statistical approaches have to be compared with respect to the single influencing factors. Differences should be explained. The taxa being sensitive towards the different influences will be identified. The inclusion of the summer data will clarify the extent of the acute effects. The inclusion of the next autumn and spring data will enable the comparison with the presented data and the quantification of the recovery. The data set of the region of Braunschweig will be evaluated in the same way to enable comparisons between the two regions and clarify possibilities of extrapolation.

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Pesticide contamination of streams and its effects on the composition of the macroinvertebrate community

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Abstract

The level of pesticide contamination of small streams is strongly influencing the species composition of the macroinvertebrate community. Samples of water taken from streams in the surroundings of Braunschweig revealed the presence of pesticide compounds in eight of the thirteen streams tested. Five streams contained only fungicides and herbicides, and in the other three streams insecticides were found as well. The communities present in highly contaminated water are dominated by insensitive species with a high potential for reproduction and hence for recolonization. In only slightly contaminated streams, sensitive species with lower reproduction and recolonization potential dominate.

Introduction

Streams are known to become contaminated by insecticides due to surface water runoff from arable land (WAUCHOPE, 1978). Whether this contamination has a significant effect on the aquatic community is under discussion. Some authors presume such an ecological significance (AUFSEß et al., 1989; HECKMAN, 1981; LENAT & CRAWFORD, 1994; SALLENAVE & DAY, 1991; TADA & SHIRAISHI, 1994). A biological response to surface water runoff contaminated with insecticides was demonstrated by the use of in situ bioassays (BAUGHMAN et al., 1989; MATTHIESEN et al., 1995). However, a firm causal connection between the contamination due to runoff and a related change of the aquatic community has been established only recently, by LIESS & SCHULZ (1999) and SCHULZ & LIESS (1999).

Each of the studies cited above was concerned with a single body of water. In the present investigation a considerable number of small streams were sampled, in order to determine the kinds of pesticides present there and to see what effects this contamination might have on the aquatic community. Small streams were selected because they represent a “worst-case scenario”: the highest pesticide concentrations have been found in such bodies of water, with catchment areas of < 100 km^2 (KREUGER & BRINK, 1988).

Methods

In the year 1998 thirteen streams were sampled. Ten of these are in the vicinity of Braunschweig (southern Lower Saxony), where a large proportion of the adjacent land was under cultivation, and three are situated northeast of Braunschweig, in the Südheide region and in Brandenburg, with no directly adjoining agricultural land.

In selecting these streams, the following minimal criteria were applied:

- They belonged to the category “lowland flowing waters”.
- They contained about the same amount of water throughout the year.
- There were no toxic inputs of non-agricultural origin (e.g., road drainage or industrial effluents).

The samples examined for pesticides were taken by an event-controlled system during runoff (LIESS et al., 1999). All water samples were fixed on C18 extraction columns (Bakerbond Polar Plus®, from the firm Mallincroft Baker). Analysis by gas chromatography and mass spectroscopy was carried out by the Institute for Ecological Chemistry at the Technical University of Braunschweig. The lower limit for detection was better than 0.05 µg/l for each substance. Few samples is still in the process of being analyzed.

To supplement the sampling of the stream community, at two-week intervals physicochemical parameters of the water were recorded, by means of electronic devices (from the firm WTW) and colorimetric quick-testing kits (Visicolor®; MACHEREY & NAGEL). The water temperature was measured by analog min-max thermometers permanently installed in the stream. To enable comparison of the morphology and characteristic structures of the streams investigated, every month during the period of the study visual observations and assessments were made of the stream dimensions, structural dimensions of the stream bed and the surrounding terrain, and how the adjacent land was being used. These values were averaged over a 100-m stretch of the stream.

The stream community was sampled every two weeks during the period from the beginning of April to the end of July, with a Surber sampler having a ground area of 0.125 m^2 and a net with 1-mm mesh. On each sampling occasion four separate samples were taken in a section of the stream about 50 m long. The samples included organisms inhabiting the sediment down to a depth of ca. 5-10 mm. Individuals with a maximal dimension less than 1 mm were omitted from the data analysis. In general, animals were identified at the species level. However, dytiscid, chironomid and simuliid larvae, as well as members of the Lumbricidae and Tubificidae, were identified only as far as the family, because more precise identification is possible only by microscopic examination after elaborate preparation.

Results and discussion

Pesticide contamination of streams

The water samples taken during 1998 from the ten streams close to Braunschweig showed that eight of these streams contained pesticides (Table 1). In five of the streams only fungicides and herbicides were found, whereas three other streams also contained insecticides at concentrations above 0.05 (µg/l). All of the documented contaminants were present in the concentration range sublethal for macroinvertebrates under acute exposure. In the three streams located northeast of the Braunschweig area, no pesticides were demonstrated.

Table 1 Maximal concentrations (µg/l) of pesticide compounds and number of times they were detected (in parentheses) in each of ten streams in the vicinity of Braunschweig during the period from mid-April to the end of July 1998. The table also gives the number of *runoff*-initiated sampling events (S) as well as the number of these samples in which pesticides were present (S+). ND: none detectable

Site	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**
Azoxystrobin	ND	ND	0.2 (2)	0.2 (2)	0.2 (2)	0.6 (4)	1.3 (5)	0.05 (1)	1.4 (3)	0.6 (10)
Bifenox	ND	ND	0.2 (2)	0.04 (1)	0.2 (1)		0.04 (1)	0.04 (1)	0.2 (2)	0.1 (2)
Deltamethrin	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
Epoxiconazole	ND	ND	0.3 (2)	0.4 (4)	0.7 (4)	0.8 (6)	0.3 (6)	0.3 (2)	0.6 (7)	0.8 (11)
Esfenvalerate	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
Kresoxim-met.	ND	ND	0.2 (2)	0.3 (3)	0.05 (1)	0.1 (1)	0.7 (3)	0.2 (2)	0.2 (4)	0.3 (7)
Lindane	ND	ND	ND	ND	ND	ND	0.02 (1)	ND	0.03 (1)	0.2 (1)
Parathion-etyl	ND	ND	ND	ND	ND	ND	ND	0.2 (1)	0.3 (4)	0.3 (2)
Pendimethaline	ND	ND	ND	ND	ND	ND	ND	ND	0.04 (1)	ND
Propiconazole	ND	ND	ND	ND	ND	0.1 (1)	ND	ND	0.7 (1)	0.08 (3)
S	**4**	**7**	**6**	**9**	**10**	**7**	**7**	**3**	**9**	**11**
S+	**0**	**0**	**3**	**4**	**5**	**6**	**6**	**2**	**7**	**11**

Pesticides were detected frequently in those streams that have an agricultural catchment. The ones most often present were the fungicides epoxiconazole, kresoxim-methyl und azoxystrobin. They were found in samples from all the pesticide-contaminated streams. Other studies also established that pesticide contamination of small headwater streams in agricultural catchment areas is marked by transient peaks (KREUGER, 1995; WILLIAMS et al., 1995). These inputs are caused by heavy rain events, inducing surface run off. Such processes (GHADIRI & ROSE, 1991) contribute considerably to the stream contamination.

Effects on the composition of the macroinvertebrate community

In ecotoxicology, relationships between stressors and biological reactions to them are generally investigated by test systems at various levels of complexity (CAIRNS Jr. & MOUNT, 1990). It is relatively rare for such investigations to be based on findings in the field, although the way natural communities are affected ought to be the ultimate criterion for such evaluations (KOEMAN, 1982). A problematic aspect of field studies is the diversity of environmental factors, the varied effects of which usually tend to obscure the ecotoxicological relationships (BUIKEMA Jr. & VOSHELL Jr., 1993).

An attempt to evaluate the present data with reference to individual species showed that for many of the streams it is essentially impossible to describe the dependence on degree of pesticide contamination at the species level. The reason is that the presence or absence of a species is also determined by a number of other environmental factors, with respect to which the streams in question differ from one another. Therefore it is preferable to base a description of the way the stream community is altered by pesticide contamination on more general, ecological classifications of the species into appropriate groups. As a foundation for such a grouping of the macroinvertebrate taxa, the following physiological and autecological characteristics are used:

The **physiological sensitivity** of the species with respect to pesticides. Because very little suitable toxicological information is available for individual species, for the present the classification is kept at the level of taxonomic order and of the nature of the respiratory system (gill-, skin- or air-breathing). Species that breathe atmospheric air directly or through a "physiological gill" make contact with the surrounding water over a relatively small surface, and hence are considered to be less sensitive to brief contamination than gill- and skin-breathers.

The reproductive potential in the form of **duration of the generation cycle**. This parameter, according to which species are categorized as reproducing semiannually, annually or biennially, is thought to have a crucial influence on a species' capacity for recolonization following catastrophic events. For instance, after a pesticide-contamination event with lethal consequences semiannual species (more than one generation cycle per year) can expand into the affected habitat relatively quickly. Biennial species take longer to restore their population size and hence would be expected to have relatively low abundances in bodies of water that are regularly contaminated by pesticides.

The 13 streams studied here were subdivided into four groups according to the land use of the catchment and the extend of their pesticide contamination, as follows.

- Streams with no agriculture in their catchment area and no evidence of pesticide content (here termed "reference stream").
- Streams with a catchment region that includes agricultural areas, but in which no pesticides were found during the study period (termed "uncontaminated").
- Streams in which fungicides and/or herbicides were detected, but no insecticides (termed "slightly contaminated").
- Streams in which, in addition to fungicides and/or herbicides, insecticides were detected (termed "highly contaminated").

Dependence of sensitivity of the species on pesticide contamination

The proportion of taxa classified as physiologically sensitive, as a percentage of the overall abundance, was significantly lower in the highly contaminated streams than in the uncontaminated and reference stream (Fig. 1).

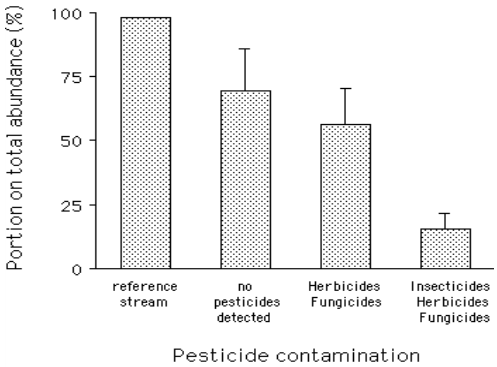

Figure 1 Percentage of the overall abundance that is represented by sensitive species, for 13 streams differing in degree of pesticide contamination. Error bars show standard error

Dependence of generation-time distribution on pesticide contamination

The proportion of annual species, as percentage of the overall abundance of macroinvertebrates, does not differ significantly in streams with different levels of contamination. However, the proportion of semiannual species is significantly greater in the contaminated than in the uncontaminated and reference streams, whereas the proportion of biannual species is significantly lower in the contaminated than in the uncontaminated and reference streams (Fig. 2).

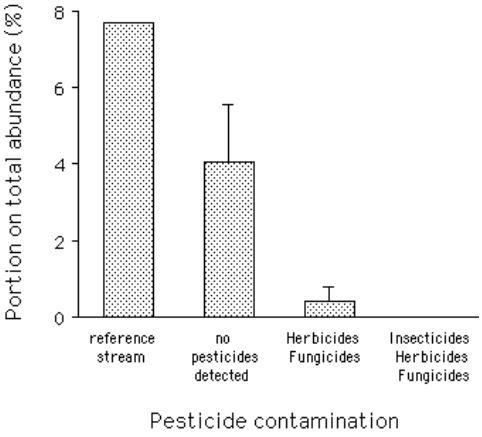

Figure 2 Percentage of the overall abundance that is represented by biannual species, for 13 streams differing in degree of pesticide contamination. Error bars show standard error

The results show that of the streams studied here, those with high pesticide contamination are dominated by macroinvertebrate communities comprising insensitive species with a high reproduction potential and hence a high recolonization potential. In streams that are only slightly contaminated, the dominant species are sensitive and have a lower potential for reproduction and recolonization. The distribution of species described here is consistent with observations at a stream, the community of which sustains significant lethal damage only during rare pesticide input peaks. With contamination peaks of this kind, however, recovery was already observed in the following year (LIESS & SCHULZ, 1999). Hence even in streams rarely contaminated by pesticides, communities with pesticide-tolerant species having a high reproduction potential are at an advantage.

Nevertheless, the reaction of the species actually observed in the field is stronger than would be predicted using the limited available laboratory toxicity data. There are several possible explanations for the difference of the concentration-response relationship in the field vs. the laboratory. It might be that the duration of contamination in the field is longer than assumed, or that the elevated contamination of suspended particles increases the overall exposure of the organisms. Another possibility is that a given species is more sensitive under field conditions than in laboratory tests. Similar findings have been described by MATTHIESEN et al. (1995), who observed 100 % mortality of caged *Gammarus pulex* following a short term peak concentration of 27 µg l^{-1} Carbofuran, which exceeded the 24 h LC50 of 21 µg l^{-1} only for a period of 3 to 5 hours. Baughman *et al.* (1989) expected differences in measured and real exposure concentrations to be a reason for higher mortalities in in situ bioassays than predicted from laboratory data.

Significance of other environmental factors

Contamination by pesticides is only one of many factors that can affect the stream biocenosis. Other environmental factors can mask the action of pesticides. For instance, investigations currently underway are revealing that communities in bodies of water that are drying out, or are severely contaminated by organic or inorganic compounds (e.g., sewage plants), undergo structural modifications similar to those following exposure to pesticides. Communities in waters negatively affected by other environmental factors would thus be expected to be less altered by the addition of pesticide contamination.

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Risk assessment and mitigation measures for pesticides: Are all patches of freshwater habitat equal?

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Introduction

When evaluating studies to assess ecological risks of pesticides, and when implementing risk mitigation measures, it is important to have scientifically sound and broadly accepted ideas of what constitutes an ecologically important effect. In other words we need to know what targets we are protecting, and whether these targets need to be the same for different types of freshwater habitat. The present paper aims to stimulate the discussion on the "what if" and "so what" questions when assessing and mitigating ecological risks of pesticides in surface water. Such a discussion is necessary to develop a broadly accepted framework that can be used to address the "ecological significance" of predicted effects and the risk mitigation measures that ideally should be implemented.

Guidance how to deal with risks of pesticides in the environment is not only provided in a regulatory context by the Uniform Principles, but also by other concepts based on science and ethics. To sharpen the discussion, three completely different approaches to manage ecological risks of contaminants can be recognised, viz.:

- the Pollution Prevention Principle,
- the Carrying Capacity Principle, and,
- the Functional Redundancy Principle.

The pollution prevention principle

The Pollution Prevention Principle presupposes that all environmental pressure is potentially harmful. Precaution is necessary since multiple-stress due to the presence of low levels of more than one chemical, or unexpected effects of metabolites (e.g., hormone disruption), can never be excluded. This opinion is in line with the community conditioning hypothesis (MATTHEWS et al., 1996), which states that ecological communities tend to preserve information about every event in their history, including stress by pesticides. It is also in line with the Rivet hypothesis (EHRLICH & EHRLICH, 1981) that presupposes that each loss of a species affects to a certain extent ecosystem integrity.

The Pollution Prevention Principle considers the "what if" question more important than the "so what" question. Consequently, emission of pesticides to non-target sites should be prevented as much as technologically and social-economically feasible, even when the criteria of the Uniform Principles are met. In its most extreme form, implementation of the Pollution Prevention Principle will restrict the use of agricultural pesticides as much as possible, e.g., by putting high taxes on the use of these chemicals, by supplying pesticides on prescription only (curative use when no other options to protect crops are available), and by subsidisation of integrated or biological farming practises.

The Pollution Prevention Principle may also be implemented in a more focussed and pragmatic way, e.g. by protecting specific non-target habitats that are considered vulnerable and of special value. An example is the protection of freshwater habitats and the coercive instruction in several European countries to apply emission reducing measures (e.g., buffer zones, wind breaks) in case the agricultural fields border surface water.

A certification system for farmers may help to implement the Pollution Prevention Principle. Such a certification system can be used to reward those farmers (e.g., by lower taxes) that apply good environmental practice, such as the implementation of buffer zones and the use of the least damaging pesticides that are available on the market.

The carrying capacity principle

The Carrying Capacity Principle presupposes that the environment can absorb and endure a certain amount of pollution. The pollution should be limited to a level before which unacceptable and irreversible adverse impacts occur on ecosystem structure and functioning. From a scientific point of view, periodically occurring stresses can be considered a normal phenomenon in ecosystems. Within limits, these factors even

may be a driving force for the development of a more diverse community. According to HANAZATO (1998) results of experimental pond studies suggest that sometimes species richness increases due to insecticide stress. This may be explained by the fact that in more stable environments fewer species co-exist due to competitive exclusion. In the course of the evolution organisms developed a large variety of strategies to survive and cope with temporal unfavourable conditions like desiccation, flooding, temperature shocks, shading, oxygen depletion, food limitations, toxins in food etc. In some cases, but certainly not all, the stress caused by a pesticide may more or less resemble that of a natural stress factor.

A conservative approach in line with the Carrying Capacity Principle is to consider a certain level of a pesticide acceptable if the sensitive structural or functional endpoints of the community are not, or only shortly, impacted. In this approach the maximum permissible concentration may be set at the critical ecological threshold level as assessed by an adequately performed micro/mesocosm test, or by the first-tier criteria of the Uniform Principles.

A liberal approach in line with the Carrying Capacity Principle is to consider a certain impact on ecosystem structure or functioning acceptable in case it is a local phenomenon and recovery takes place within a realistic time-frame (days to weeks, dependent on the life-cycle characteristics of the affected species). When environmental stress is restricted in space and time, and non-polluted habitats are nearby, recovery processes may guarantee the sustainability of sensitive populations. According to BROCK and BUDDE (1994) recovery of aquatic populations from pesticide-stress may be rapid if the compound is not persistent or when its bioavailability rapidly decreases, the generation time of the populations affected is short and/or if there is a ready supply of propagules of eliminated populations.

DOMSCH et al. (1983) suggested to use the “normal operating range” of population densities and functional endpoints in specific ecosystems as a baseline against which to assess pesticide-induced changes. In other words, effects of pesticides that are restricted in space an time may in certain habitats be regarded as unimportant when they are of a smaller scale than changes caused by other natural or anthropogenic stressors.

The functional redundancy principle

The Functional Redundancy Principle presupposes that for a sustainable functioning of an ecosystem a decrease in biodiversity needs not to be dramatic, at least when key species are not impacted. This because of the redundancy in roles and functions provided by the surviving species in the community. This principle is in line with the Functional Redundancy hypothesis (LAWTON, 1994).

When adopting the Functional Redundancy Principle the emphasis is on ecosystem processes; impacts are considered acceptable when functional attributes are not changed, despite possible effects on community structure. Functional endpoints integrate the effects on lower levels of biological organisation, and indicate the severity of a perturbation.

According to KERSTING (1994) functional endpoints are rarely more sensitive than structural ones (exceptions are photosynthesis inhibiting herbicides). Effects on functional endpoints indicate the limit of functional redundancy within the stressed community. Once ecosystem processes have changed due to pesticide contamination, this usually is an indication of really severe effects on structural endpoints and that the stressed ecosystem really is “sick”. Unfortunately it cannot be stated that the ecosystem is healthy as long as the functions have not changed.

Are all patches of freshwater habitat equal?

The goal of ecological risk assessment and risk mitigation measures is to contribute to sustainable management of freshwater habitats. To keep or restore wetlands and aquatic habitats in a more or less pristine condition is important for areas designated as nature reserves. For freshwater ecosystems in areas with intensive agriculture this is practically not feasible, but in many cases we aim these systems to be multifunctional.

An important question is whether different types of freshwater ecosystems also differ in sensitivity. For this it is important to address the system-specific differences in fate of pesticides, the critical threshold levels for effects, and the conditions that might stimulate or hamper recovery. (Semi-)field and model studies have demonstrated that several factors may have a large influence on pesticide-behaviour in freshwater ecosystems such as water flow (dilution), the pools of dead or living organic matter present (e.g.

macrophytes may effectively sorb a large proportion of dose), water depth (sorption to the sediments is usually slower in deeper water) and physical-chemical properties (pH may have a large impact on hydrolysis of the compound). Nevertheless, in case of a more or less similar exposure regime to the same compound, results of aquatic micro/mesocosm tests performed with herbicides and insecticides suggest that critical threshold levels do not differ much between studies performed in different types of test system and in different regions (VAN DEN BRINK, 1999; VAN WIJNGAARDEN & BROCK, 1999 and literature cited therein). The recovery-rate of sensitive populations, however, considerably varied between studies. This depended on the degree of isolation of the test systems and the life-cycle characteristics of the populations present. In general, species with a relatively short generation time, resistant life stages and/or well developed abilities to disperse (e.g. *Algae*, *Daphnids*, multivoltine insects) showed a fast recovery in contrast to organisms with a more complex life-cycle or with a low ability to recolonize isolated systems (e.g., *Gammarus*, univoltine insects).

Since properties of freshwater habitats within the agricultural landscape vary in space and time it is important to have scientifically sound and broadly accepted ideas of what constitutes an ecologically important effect and what targets we aim to protect (see e.g., BROCK & RATTE, 1999). Also the question whether all patches of freshwater habitat should be protected equally needs to be addressed. For this we need clear reference and target images for freshwater habitats potentially at risk in a certain region. Reference images are based on general knowledge of the structure and functioning of the different types of freshwater ecosystem that occur in a certain region. The target image can be regarded as the embodiment of what we consider feasible risk assessment goals, and they are the result of scientific, political and social discussions (LENDERS et al., 1998). In the context of aquatic risk assessment for pesticides, and when implementing risk mitigation measures, we need at least clear reference and target images of aquatic ecosystems that are common in agricultural landscapes, such as ponds, shallow lakes, drainage ditches and streams.

A case on drainage ditches in the Netherlands

Drainage ditches are very typical elements of the Dutch landscape. The total length of ditches in the Netherlands is calculated to be no less than 300.000 km (HIGLER, 1989). The main function of these ditches is the regulation of the water table to allow agriculture and human occupation. However, also the importance of ditches as suitable habitats for water organisms is recognised. The policy in the Netherlands is to consider drainage ditches as multifunctional elements of the landscape. Characteristic features of drainage ditches are their typical dimensions (shallow, line shaped, immense network), that they are man-made and usually highly influenced by agricultural practices, the usually small water flow, and last but not least, the annual management regime consisting of removal of aquatic vegetation and sludge to maintain the drainage capacity.

Community structure in drainage ditches may vary considerably in space and time. Species richness of the invertebrate community usually is relatively high in larger permanent ditches when a structurally diverse macrophyte vegetation is present. In contrast, the diversity of invertebrate populations is relatively low in the smaller and shallower temporary ditches. Small ditches with simpler vegetation structure or organically enriched sediments have an intermediate position (VERDONSCHOT & HIGLER, 1989; HIGLER & VERDONSCHOT, 1989). Particularly in the smaller ditches the community present may predominantly consist of populations with a short generation time (r-selected species) due to environmental conditions like temporal desiccation, pulses of nutrients and manure, variable water flow and mechanical cleaning of vegetation. In these systems temporal pesticide-stress most probably will result in less severe impacts than in the larger, permanent ditches and streams, also characterised by species with more complex life-cycles and a longer life-span and generation time (e.g., univoltine insects).

Because of the abundant presence of drainage ditches in the Dutch landscape it is difficult to respect the Pollution Prevention Principle everywhere. For this reason the Dutch Ministry of Agriculture, Nature Management and Fisheries proposed a pragmatic policy for differentiated hazard evaluation of pesticides in surface waters (ADRIAANSE et al., 1997). This policy presupposes that differences in functionality of the non-target habitats at risk may be used to differentiate between water quality criteria and risk mitigation measures. The proposed policy was based on the following elements:

- Differentiation in water quality standards between smaller field ditches and main water courses.

- Temporal exceedance of water quality standards is allowed in smaller field ditches in case the community recovers within a few months.
- In main water courses no exceedance of water quality standards (according to the Uniform Principles) is allowed.
- Along main water courses buffer zone will be established.
- Monitoring results of main water courses indicating that the 90 percentile in time and space of measured concentrations exceed water quality standards will be coupled back to registration.
- To date no political consensus is reached in the Netherlands on the proposed policy described above. There is, however, an increasing willingness to consider differentiation in protection level and risk mitigation procedures to address environmental problems.

Guidance for risk mitigation measures

When aiming to protect aquatic habitats from pesticide-stress, and consequences of effective risk mitigation measures are economically and socially acceptable, the wisest decision is to adopt the Pollution Prevention Principle. Financial resources, however, usually are limited and the different stakeholders certainly have different interests.

A differentiation in protection level of aquatic habitats that takes into account perceived differences in ecology and functionality of surface water, may contribute to a more focussed implementation of risk mitigation measures. This may secure an adequate protection of at least to most vulnerable and valuable habitats. A choice for the Pollution Prevention Principle seems in all cases logical for patches of aquatic habitat in areas assigned as nature reserves. An approach in line with the Carrying Capacity Principle might be adopted for non-target habitats such as multifunctional surface waters in the agricultural landscape. The Functional Redundancy Principle seems appropriate for agricultural target sites only, e.g. aquatic systems assigned to have predominantly an aqua-culture function such as fish breeding ponds, rice fields or commercial water-cress beds.

When developing a framework in line with the Carrying Capacity Principle, and that can be used to address the “ecological significance” of predicted concentrations of pesticides in freshwater habitats bordering agricultural fields, the following arguments seem to be important:

- We need clear reference and target images of different types of aquatic ecosystem in agricultural landscapes that can be used to identify the valuable freshwater habitats present in the area of interest.
- Ideally, risk mitigation measures should be implemented that protect these valuable freshwater habitats from exposure to pesticide concentrations above the critical threshold level (conservative approach in line with the Carrying Capacity Principle).
- To achieve sustainable freshwater populations, small isolated aquatic ecosystems like ponds need stricter risk mitigation measures (e.g. wider buffer zones) than interconnected watercourses (because of differences in recovery rates). Similarly, more or less permanent and structurally diverse aquatic habitats (e.g. streams and larger drainage ditches) need stricter risk mitigation measures than field ditches that regularly fall dry or suffer other stresses (e.g. frequent removal of vegetation).
- Stricter risk mitigation measures are advocated for upper reaches of streams (sources for recolonization), waterways that supply water to nature reserves, and aquatic systems influenced by seepage (potentially diverse freshwater community).
- In all multifunctional freshwater habitats of agricultural landscapes, possible side-effects of pesticides should be reversible and smaller than the fluctuations caused by other natural and anthropogenic stresses normal for these systems. In addition, these potential side-effects should always be restricted to an acceptable spatial scale, and water transport of pesticides to more protected aquatic habitats should be negligible (liberal approach in line with the Carrying Capacity Principle).
- Within agricultural landscapes enough refugia should be present or maintained to allow recolonization of affected populations.

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Terrestrial environment

Protecting field edges and boundaries from pesticides: the benefits for farmland wildlife

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Introduction

Since the 1950's many forms of wildlife associated with agricultural ecosystems and particularly cereals have declined throughout Europe. Most effected have been the birds (Tucker & Heath, 1994), butterflies (Heath et al, 1984), beneficial insects (Aebischer, 1991), annual arable wildflowers (Schumacher, 1987) and game species, especially the grey partridge, *Perdix perdix* L. (Potts, 1986). These declines have coincided with the intensification of agriculture driven by the EU policy for European self-sufficiency in food production, food security and maintenance of farm incomes. The changes in agricultural production systems which have had the most influential effects on wildlife in the UK are the increase in field achieved by removing hedgerows, other boundary cover and non-crop cover types so that fields have become larger and the diversity of the landscape mosaic less complex; rotations have become less complex; mixed farms have been replaced by specialised farms and even specialised regions of production have developed; artificial fertilisers and pesticides have replaced the need for husbandry practices and winter cropping has expanded so deriving seed-eating birds of feeding grounds.

Cereal fields are not yet, however, deserts derived of wildlife but if properly managed can harbour a vast variety of farmland wildlife. In Europe up to 700 species of plants have been recorded in cereal fields and in the UK alone 2000 species of insects and spiders are found (POTTS, 1991). This may still be less than that found in nature reserves but is still important because of the area of land occupied by cereals, currently 17% of the UK.

Grey partridges as a case study

The grey partridge has acted as barometer of changes in agriculture because its decline has been linked to agricultural intensification. In order to identify the factors contributing to the decline of wild breeding birds, the Game Conservancy Trust has monitored annually since 1968 invertebrates and weeds in approximately 100 fields across West Sussex. From this study and others in became apparent that the key factor causing changes in grey partridge populations was the level of chick survival. This was associated with the availability of sufficient quantities of preferred insects (Southwood & Cross, 1969; Green, 1984; Hill, 1985; Potts, 1986). More recently survival of corn bunting (Brickle, 1997) and skylark (Poulsen *et al*, 1998) chicks has been linked to availability of insects. The main factors causing the decline of insects preferred by chicks were the use of pesticides (insecticides, herbicides and fungicides) (Potts, 1986) and the decline of undersowing to establish a grass ley after cereals. Five main groups of insects have been identified as being important dietary components for grey partridge chicks. Insects included a group of *Coleoptera* (*Chrysomelidae*, small diurnal *Carabidae* and *Curculionidae*), larval forms of *Lepidoptera* and *Tenthredinidae*, and many members of the *Heteroptera* (especially of the genus *Calacoris*). More recently *Coleoptera* (especially *Carabidae*), *Orthoptera* and larvae of *Lepidoptera* were linked to the survival of farmland birds in general (Campbell *et al*, 1997). The latter two are dependent on permanent grass and perennial herbs for their survival.

Insecticides applied in spring and summer to control aphids and other cereal pests were identified as being the most harmful to the chick-food insects (Vickerman & Sunderland, 1977; Sotherton, 1990; Moreby & Sotherton, 1997). More recently attempts to control the orange wheat blossom midge, *Sitodiplosis mosellana* Géhin, have resulted in the widespread application of the broad-spectrum compounds chlorpyrifos and triazophos. More harmful in the long-term has been the increase in use and effectiveness of herbicides. Four of the grey partridge chick-food groups are dependant on broad-leaved weeds (Chiverton & Sotherton, 1991; Sotherton, 1991; Sotherton & Moreby, 1992) and, in the case of saw fly caterpillars, grasses (Vickerman, 1974); they are thus affected by herbicides. Preventing spray drift into field margins is important if *Lepidoptera*, *Symphyta* and *Orthoptera* are to survive in these areas.

Development of conservation headlands

Grey partridge and pheasant chicks forage for insects and weed seeds predominantly along the edges of cereal fields (Green, 1984; Hill, 1985). Many of the chick-food insects abound at the edges of cereal fields therefore one solution was to selectively spray the cereal crop field margins thus the Conservation Headland was devised (Rands & Sotherton, 1992; Sotherton, 1991). This had the benefit of allowing farmers to maintain high production levels while ameliorating some of the observed effects of pesticides on farmland wildlife. In this management system, the outermost section of the spray boom (in most cases the outermost 6 m) was either switched off when spraying around the crop edges, or the headlands were sprayed separately with more selective compounds. Screening trials were carried out to determine which pesticides could be applied to Conservation Headlands and guidelines have been produced for farmers by The Game Conservancy Trust. Results of the screening trials have been published in part elsewhere (Rands, 1985a; Sotherton *et al*, 1993). A number new insecticides (tau-fluvalinate, zeta-cypermethin, a deltamthrin/pirimicarb mix) have recently become available for application to cereal crops and these along with dimethoate, deltamethrin and pirimicarb are currently under evaluation for their toxicity to chickfood insects. Preliminary results of a replicated field trial indicate that pirimicarb and tau-fluvalinate are less toxic to chick-food insects, but there is little difference between the other insecticides. Similar results were found when sawfly larvae (Tenthredinidae) were fed *Poa annua* leaves which had been sprayed in the field.

Benefits of conservation headlands

The technique has been extensively tested within the UK, Sweden, Holland and Germany and the benefits have been shown for grey partridges (Rands, 1985b), wild pheasants (Sotherton *et al*, 1993), arthropods important as chick food (RANDS, 1985a; CHIVERTON & SOTHERTON, 1991; MOREBY & SOUTHWAY, 1998), predatory insects (deSnoo *et al*, 1995; deSnoo & deLeeuw, 1996), butterflies (RANDS & SOTHERTON, 1986; DOVER, 1991; DE SNOO & DE LEEUW, 1996;), hoverflies (Cowgill *et al*, 1993b), arable weeds (SOTHERTON, 1991; WILSON, 1994a) and small mammals (Tew *et al*, 1992).

Gamebird chicks

Significant increases in the survival of grey partridge chicks were found in cereal fields surrounded by Conservation Headlands with a mean percentage chick survival rate of 39.1% compared to 23.3% for fully sprayed fields during the period 1984-1991 (SOTHERTON, pers. comm.). Similar increases in pheasant chick survival was also found where Conservation Headlands had been used (Sotherton & Robertson, 1990; Rands & Sotherton, 1992).

Other farmland birds (such as the corn bunting and skylark) undergoing decline which rely on insects as chick-food, may also benefit from Conservation Headlands. Three farmland birds foraged more in Conservation Headlands than in those which had been sprayed (deSnoo *et al*, 1994).

Invertebrates

A number of studies have shown that Conservation Headlands support higher densities of chick-food insects (RANDS, 1985a; CHIVERTON & SOTHERTON, 1991; MOREBY & SOUTHWAY, 1998). A three year survey of winter wheat during May to July revealed that the density of most chick-food insects was higher in Conservation Headlands compared to the sprayed crop and that it was the presence of weed cover per se that was important rather than any individual weed-insect species relationships (MOREBY & SOUTHWAY, 1998).

Butterflies have also been shown to benefit from Conservation Headlands. In a five year survey of field margins adjacent to Conservation Headlands, between two and four more species and more individuals of each species were observed compared to margins which were fully sprayed (Dover, 1991) and 68% of all butterflies observed were in Conservation Headlands (Dover, 1994). Weeds within the Conservation Headlands are mainly used for their nectar with *Sinapsis arvensis*, *Cirsium arvense* and Compositae being favoured. Some may act as larval host plants, for example *S. arvensis* to some of the Pieridae (Dover, 1989). In addition, Conservation Headlands can act as buffer zones preventing insecticide drift into the adjacent hedgerow (Cuthbertson, 1988; Longley & Sotherton, 1997c). The level of drift is also mediated by the wind speed, crop and height of hedgerow flora. Where the outer 6m is unsprayed, the deposition in the hedgerow increases much less with increasing weed speed (Longley & Sotherton, 1997c). When foliage from field margins with and without Conservation Headlands was compared, mortality of *Pieris brassicae* larva was reduced by almost 90% (Longley *et al*, 1997a).

If herbicides reach the flora at the base of hedges they may cause sufficient damage to change the species composition and thereby the phytophagous insect community or at lower rates prevent flowering and deprive pollinators of food (Boatman *et al*, 1994a). Similarly, misplaced nitrogen fertiliser may also alter the nutrient balance and allow nitrogenous annual weeds to flourish (Boatman *et al*, 1994b). The physical structure of the hedgerow flora is also important especially for over wintering species and changes to this by herbicides or fertilisers may alter its suitability.

Other groups of beneficial insects may also exploit the resources provided by Conservation Headlands. Hoverflies (Syrphidae) make use of the nectar and pollen provided by flowering weeds (Cowgill *et al*, 1993a). Bumblebees, another group which is undergoing a serious decline in farmland, utilised the weed flowers, mainly poppies (*Papaver rhoeas*) within Conservation Headlands before the perennial plants in the hedgerow started to flower of which knapweed (*C. scabiosa*) was the most attractive (Fig. 1).

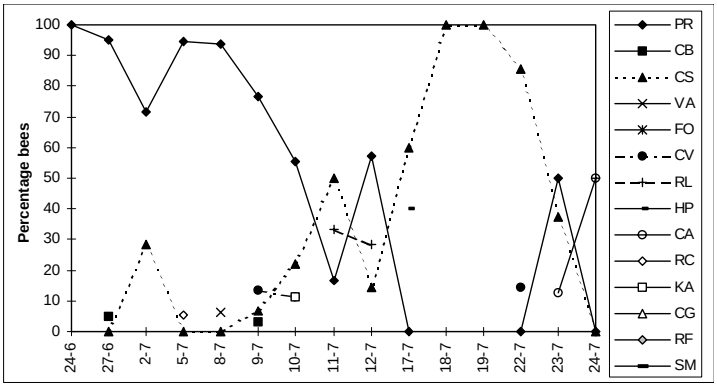

Figure 1 Variation through time in usage by bumblebees of the plant species in Conservation Headlands and the adjacent hedgerow. (*PR = P. rhoeas, CB = C. bulbosum, CS = C. scabiosa, VA = V. arvensis, FO = F. officinalis, CV = C. vulgare, RL = R. lutea, HP = H. perforatum, CA = C. ascendens, RC = R. canina, KA = K. arvensis.*)

Higher numbers and in some cases greater diversity of *Carabidae* were found in the unsprayed edge compared to an edge sprayed using the standard farm practice (deSnoo *et al*, 1995). Polyphagous predators have also been found to be better fed within Conservation Headlands and this may lead to their increase through improved diet and fecundity (Chiverton & Sotherton, 1991).

The majority of studies evaluating the impact of insecticide drift into non-target areas focus on one or two species which mostly found in the hedge base. However, the hedge also harbours a higher number and diversity of insects. An insecticide fogging technique found 13 orders and approximately 80 invertebrate families present within a typical mature hedge in southern England (Joyce *et al*, 1997) with 40 individuals m^2. Many of these families had not returned to their pre-treatment numbers 55 days after spraying and the relative composition of the different functional groups changed (Joyce, 1998). This implies that insecticide drift, if of sufficient level to cause mortality or sub-lethal effects may similarly disrupt the ecology of non-crop habitats.

Weeds

Approximately 200 plant species are found in arable crops in the UK (Wilson, 1994) but many have declined in their distribution since the advent of intensive farming and over a quarter are threatened with extinction (Wilson, 1994). Many more weed species are found around the edges of arable fields, their numbers and diversity progressively declining with distance from the field margin (Wilson & Aebischer, 1995). Thus it was not surprising that the implementation of Conservation Headlands was shown to encourage their survival. A substantial increase in weed cover, species diversity and biomass was found in Conservation Headlands (Wilson, 1989; Sotherton, 1991). Besides herbicide use, other factors such as crop type, nitrogen fertiliser inputs and crop sowing date can influence weed germination and survival (WILSON, 1994a).

Small mammals

Wood mice were shown using radio tracking to forage more in Conservation Headlands, the greater abundance of grass weed attracting them and they were observed feeding on the blackgrass seeds (Tew *et al*, 1992).

The spatial distribution of insects

Several studies have demonstrated that beneficial species including Carabidae (Hengeveld, 1979; Thomas *et al*, 1998), parasitic wasps (Ruggle & Holst, 1995; Longley *et al*, 1997b), Linyphiidae (Thomas *et al*, 1990) and pest species such as cereal aphids (WINDER et al., 1998) are not homogeneously distributed within fields or even within the landscape, but exist as a series of local semi-autonomous populations (DEN BOER, 1990), termed a metapopulation (HANSKI, 1994). These populations interact through dispersal, thus the size of each metapopulation and the distance between each will be governed by their dispersal ability and the distribution of appropriate habitat. In addition to spatial heterogeneity, insects also exhibit temporal variability in abundance and activity, and all of these interact thereby creating difficulties when trying to interpret the impact of pesticide applications.

How insects are distributed has implications for the design and interpretation of pesticide trials. Insecticide effects may be difficult to interpret or fail to detect the true effect if only small areas within a field are monitored and a species exhibits low numbers and spatial heterogeneity (Mead-Briggs, 1998). For example, in large-scale farming system studies the centre of a plot or a field is usually monitored using ~10 sampling locations (Holland *et al*, 1994). However, this approach sometimes fails to detect the expected reduction following the application of an insecticide with a high toxicity to beneficial species (Büchs *et al*, 1997; Frampton, 1998; Holland *et al*, 1998). This may be because the variability in insect densities is greater between sampling areas than that imposed by any farming operation. In addition, the taxonomic level at which results are analysed is also important; no effect may be detected at the family level but individual species may vary considerably in their response (Büchs *et al*, 1997). To some extent the above factors can be overcome by using a replicated plot approach as is recommended for insecticide trials (Brown *et al*, 1990; Anon, 1991), but if a species is absent or present only in low numbers within some of these areas the replication is reduced. This was clearly demonstrated in a series of four field trials conducted in the UK. Of the 28 statistical comparisons only five carabid species indicated a significant treatment effect for dimethoate (Mead-Briggs, 1998), despite its known toxicity to this family (Floate *et al*, 1989; Cilgi *et al*, 1996). Unsprayed control areas are also usually present within the field and reinvasion from these can be greater than the field margins (Duffield & Aebischer, 1994). Moreover, the plot size in field trials rarely exceed 1 ha and reinvasion is more rapid than would occur in larger fields, thus the duration of impact is dependant upon the size of the treated area (Duffield & Aebischer, 1994).

By sampling across whole fields the influence of spatial heterogeneity may be overcome and the trial represents what happens on farm, however, unless many fields are treated a different approach to statistical analysis is needed. The recently developed SADIE (**S**patial **A**nalysis by **D**istance **I**ndic**E**s) techniques can be used to detect and measure the degree of spatial pattern in spatially-referenced count data (Perry, 1998a; Perry *et al*, 1999). These were developed specifically for clustered ecological count data, in which patterns are dynamic and patchy, a large proportion of zero counts occurs frequently, and abundance has a non-stationary covariance structure. Such data are inappropriate for geostatistical analysis. The methods make full use of the data concerning each count and the two-dimensional co-ordinates of the sample unit at which it was taken; there is no restriction on the location of the sample units. The first method (Perry, 1998b) provides indices to indicate the overall level of aggregation. PERRY et al. (1999) have extended these methods to provide an index of clustering for each of the sample units, to measure the degree to which the observed count at each unit contributes to this overall aggregation. Furthermore, the clusters are identified separately, either as patches within which neighbouring units have counts relatively larger than the mean, or as gaps within which neighbouring units have relatively small counts. The SADIE class of techniques were also augmented by PERRY (1998b) to provide indices and tests of spatial association for the case when two separate arrangements (e.g., different species) share the same sample units. The software is freely available from its author. However, as with replicated plot trials, for analysis there is a lower limit on the number of an individual species which must be captured. The most recently developed of these techniques (Perry *et al*, 1999) was used to examine insect distributions before and after an insecticide application.

Insecticide applications may disrupt these distributions, the extent of which will depend on the size of the metapopulation with respect to the area treated and whether any barriers exist which could prevent reinvasion (Sherratt & Jepson, 1993). Reinvasion by *Carabidae* and *Linyphiidae* was previously shown to occur predominantly from the field margins inwards (Jepson & Thacker, 1990; Thomas *et al*, 1990), thus where Conservation Headlands are used reinvasion of treated fields from these unsprayed crop areas and field boundaries may be quicker and more extensive than in fully sprayed fields. To test this the distribution of invertebrates consumed by farmland birds was determined within a cereal field using a grid of pitfall traps (HOLLAND et al., 1999). To explore the potential of recently developed spatial analysis techniques the data was analysed using an improved SADIE method described in PERRY et al. (Perry *et al*, 1999). This technique indicates whether the arthropods were clustered, either in patches of greater than average density (indicated by indices v_i and their average value $\bar{v}_i$) or gaps were present of less than average density (indicated by indices v_j and their average value $\bar{v}_j$). Unlike previous methods, the degree of clustering is measured at each location sampled, for each of which there is an index of clustering. Values of $v_i > 1$ indicate greater patchiness and values of $v_j < -1$ indicate the presence of gaps which are larger than would be expected by chance. This revealed that these invertebrates (Fig. 2, next page) and also many individual taxa (Honek, 1988) (HOLLAND et al., submitted) were highest around the field edges and consequently these areas are the most important to protect. When an insecticide was applied across the field leaving an unsprayed 6m buffer zone around half of the margins most of the invertebrates were removed. The buffer zone did not appear to protect those individuals within the buffer zone, probably because they were at some stage moving in the treated area of the crop. Reinvasion was mainly from the field margin inwards and appeared to be more extensive in the area surrounded by the buffer zone.

Conclusions

Conservation Headlands despite their apparent benefits to wildlife have not been widely adopted. They are only used on 0.3% of the available field margins in the UK. This is because farmers perceive them as causing weed problems, although it has been clearly demonstrated that these can be avoided through careful use of selective herbicides. Any yield loss is unacceptable even though the outer edge of the crop is often the most unproductive due to competition and shading from hedgerows. They require some expertise to manage but guidelines and advice are readily available. It may be that legislation and/or compensation is needed if they are to be more widely used. The value of natural pest control as part of an integrated crop production method is now being recognised and this may assist in the uptake of Conservation Headlands. Field margins have been long recognised as important over wintering sites for beneficial species, but it may be their floral resources and value as a refuge from insecticides which will make them more attractive component of an integrated farming system.

The distribution of many farmland invertebrates appears to be heterogeneous, but with higher numbers around the field edges confirming their importance as feeding sites for farmland birds. The habitat requirements are unknown for most farmland invertebrates, although the studies on Conservation Headlands have indicted that weed cover is important. Future work will try to determine the most important abiotic and biotic factors controlling these distributions using spatially structured sampling regimes and how different boundary types differ in their ability to support beneficial insects and whether this consequently benefits pest control. A whole field application of an organophosphate insecticide caused substantial reductions in total invertebrate numbers for up to one month after spraying. Use of more selective insecticides and more precise application to the target area is advocated if beneficial invertebrates are to be preserved within fields. The inclusion of an unsprayed buffer zone appeared to encourage quicker and more extensive reinvasion after spraying but needs verification. The new SADIE technique was valuable in identifying when spatial pattern was present and the v_i and v_j indices aided the location of these areas which was not possible with previous statistical methods. The spatial variability of invertebrate distributions highlighted the importance of selecting the most appropriate sampling strategy if this variability is not to mask treatment effects.

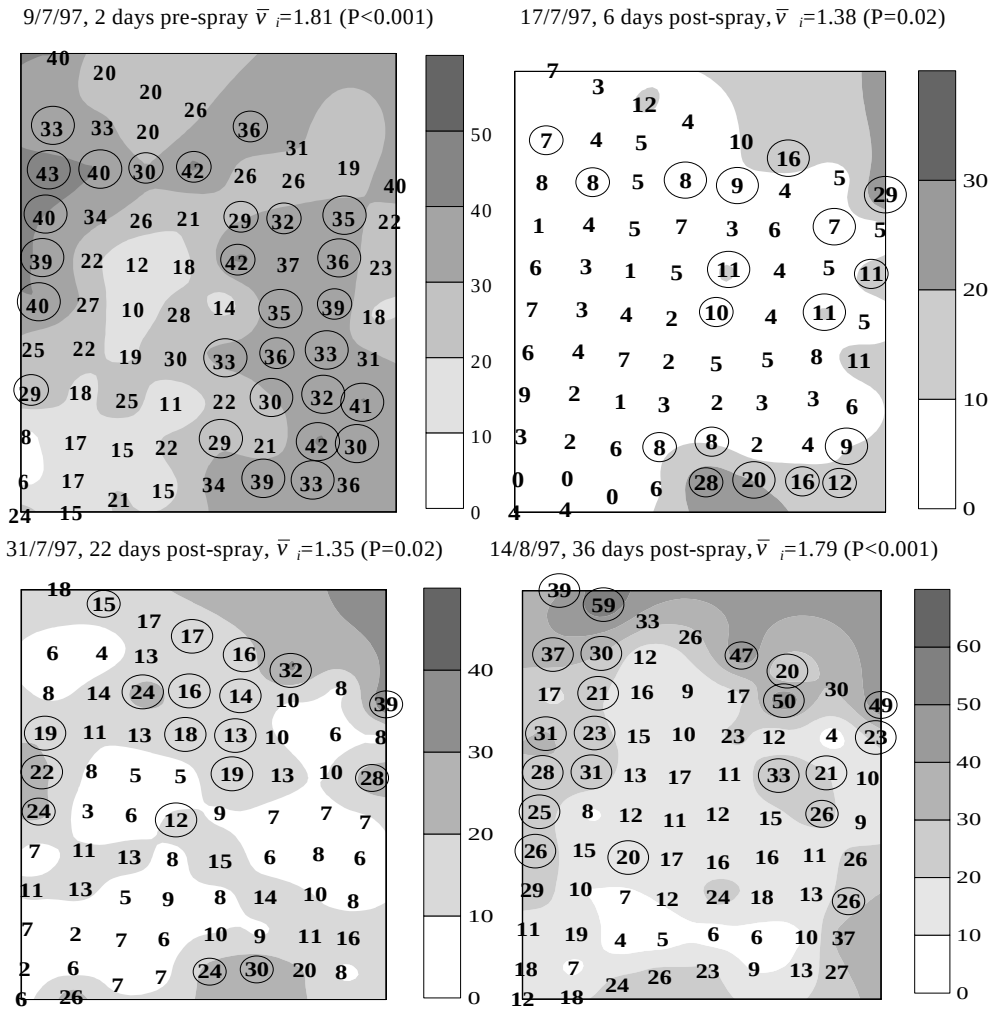

Figure 2 Arthropod counts sampled on an approximately rectangular 30m grid in a wheat field during 1997, just prior to and after spraying with dimethoate. The counts are categorised by interpolated contouring into equally spaced shaded density classes. Above-average clustering at each sample unit into patches of greater than average neighbouring counts is measured by the clustering index, v_i. Strong clustering into patches is indicated by units surrounded by larger circles with $1.5<v_I$, smaller circles indicate $1<v_i<1.5$. In each case, the average value of the patch clustering for the entire sample, $\bar{v}_i$ is shown above the map, together with its statistical significance on the null hypothesis that the observed counts were arranged randomly amongst the sample units. For clarity, the other form of clustering, into gaps of smaller than average neighbouring counts, is not shown here, but generally attains similar significance to that of the patches

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Field study on effects of insecticide applications in wheat on the arthropod community of field boundaries

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Introduction

KÜHNE et al. (1999) have calculated that in Germany the border between crop fields and field margins extend to a total length near 1.5 million km. This fact is reason enough to ask for the risk of possible side-effects of insecticide drift on off-crop habitats next to fields. While a lot of studies on the composition and population dynamics of arthropod community in different boundaries have been performed, our knowledge of side effects of insecticide treatments in crops on the non-target arthropod community in adjacent field boundaries is insufficient. Especially long-term effects, considering recovery processes of populations under worst case and more or less common conditions have not been investigated systematically. In spite of that, this important subject of applied ecotoxicology is treated already often in theoretical models for distinction and recovery processes, especially in connection with metapopulation models (SHERRATT & JEPSON, 1993; SPROMBERG et al., 1998). Experience shows that complex field studies face an enormous methodological problem and a high level of man power is needed. Considering these difficulties, an extensive and complex field study was started at the BBA branch in Kleinmachnow in 1998. The investigations should last over 3 seasons and represent a joint project of 4 BBA institutes and is realised by a working group with the experts named in the heading.

The aim of the project was to get not only seasonal regarded data on the impact of insecticide applications to wheat on arthropods in an adjacent field margin as a result of spray drift, but also an overview over possible accumulated effects of the 3 insecticide applications on the same plots during the 3 experimental seasons.

In detail data are expected to

- exposure of habitats and arthropods,
- dimension of the impact of an insecticide on beneficials and other non-target arthropods,
- effects on abundance, diversity and structure of arthropod community,
- recovery processes in arthropod populations,
- certain long-term effects and
- different evaluation of different field margin structures. However, the studies should primarily contribute to collect methodological experience and to improve the methodological approach.

In the following the authors inform about the methodological approach and some preliminary results of already completed data analyses.

Material and methods

Experimental site

The experimental site is located several km south to Berlin in a landscape with a mixture of woods and arable land. In this area only sandy, moderately fertile soils occur. For the study a 50-ha field bordered by 2 very long field margins in the west and in the east was chosen. These margins are of different types:

- a grass-dominated herb strip and
- a herb strip-hedge combination (Brandenburg stack-wood hedge).

It was decided to concentrate the activities on the 6 m wide grass-dominated herb strip because of the sufficient length of 470 m and relative uniformity of the present vegetation. In this paper only the activities on this field margin are considered. Because of the favourite conditions the strip was divided into 8 similar plots. That means to have true replications, 4 replications with and 4 plots without insecticide influence from the adjacent wheat field. Nevertheless we must calculate 2 methodological problems:

- the limited density of the arthropod community in grass-dominated herb margins and
- possible migrations and other interactions between the plots.

Insecticide application

Insecticide application should be performed as a classic measure against cereal aphids according to the rules of good plant protection practice. Karate WG (7.5 g a. i. per ha, lambda-cyhalothrin) as 300 l (400 l in 1998) spray mixture per ha was used. Spraying took place at the end of flowering (BBCH 69) using a sprayer with a working width of 18 m and a boom height of 60 cm above wheat.

Bioindicators

To assess the insecticide effects on non-target arthropods a mixture of monitoring and investigation methods has been considered. We are interested in a range of bioindicators, mainly all important beneficials and special arthropod groups like carabids, predatory flies or grasshoppers. Beside data on population density, activity, mortalities in defined populations or residues in arthropods the study should also involve data on diversity.

The monitoring methods

a) Biocoenometer surveys

This method has been chosen to obtain real densities of arthropods. Inside a randomly placed cage all plants are cut off and packed in bags, the remaining arthropods were caught by means of a suction sampler (modified vacuum cleaner). The cage has a size of 1 m². In 1998 we began with 0.25 m²-cages that were found to be too small to get minimum densities for statistical analysis. Within the margins, 4 samplings per plot and date were made, in the fields 2 samplings. The surveys in the wheat were made 3 m beside the margin and in the grass strip along lines 1 m and 5 m from the crop border. Determination of arthropods (orders, families, partly genus and species) is carried out at the laboratory after freezing conservation.

b) Malaise traps (only in 1998)

With this method, influences on the flight activity of insects were to be studied. But the results of the first year were insufficient. Therefore we left this method.

c) Pitfall traps

This widely used method should give data on the activity of polyphagous predatory arthropods including important information on the diversity under the influence of a pesticide. In each plot, one trap was placed each in the crop and the margin, 3 m away from the field border. However, in 1998 the traps stood in the wheat crop 1 m away from the field border.

d) Grasshopper cage countings

To estimate the density of grasshoppers in the grass strip, counts were carried out within 1-m² cages. The cages were randomly placed along lines 1m and 5 m from the field borders like with biocoenometer samplings. 8 samples per plot and date were taken.

e) Measurement of the contamination of field margins by spray drift

The sediment of spray on treated wheat and margins were measured indirectly using a fluorescent marker (BSF) which is added to the spray. Pipe cleaners are used to catch the spray for measurement. The pipe cleaners are placed at a distance of 1m along 3 measuring lines extending over each plot opposite to the treated wheat plot. To measure the deposit on the pipe cleaner, they are washed with water. The wash water is assessed fluorometrically.

f) Residue analysis

To assess the initial concentration and fate of the active substance, on leaves and soil, samples were taken from the field margin plots and analysed in laboratory.

g) Studies on exposure of non-target arthropods

Three different investigations were made. Firstly, assessment of insecticide contamination of carabids and spiders that were captured in dry pitfall traps and were analysed in laboratory. One trap per plot in the middle of the grass strip (3 m away from the wheat) was used. Secondly, semi-field study with reared *Poecilus cupreus* kept in low open metal frame cages (HEIMBACH et al., 1992). Cages were placed in strip and wheat

crop, on the line 1 m and 3 m line away from the field border respectively. Furthermore, grasshoppers were collected in the margin for laboratory analyses.

Time table for the samplings

The samplings on grass strip and wheat field took place 1 or 2 days before and after the insecticide treatment, and then mostly weekly. The biocoenometer surveys were performed on 4 (1998) or 3 (1999) dates. Pitfall traps were checked weekly.

Results

In both seasons spray drifted into the margin, both on the grass strip to the east and into the stack-wood hedge strip to the west where the insecticide application has been realised in both years 1 week later than at the grass strip. The results of the spray drift investigations with the fluorescent marker have shown that the drift indications decreased both with the increasing distance from wheat field and from top to bottom of vegetation (Fig. 1).

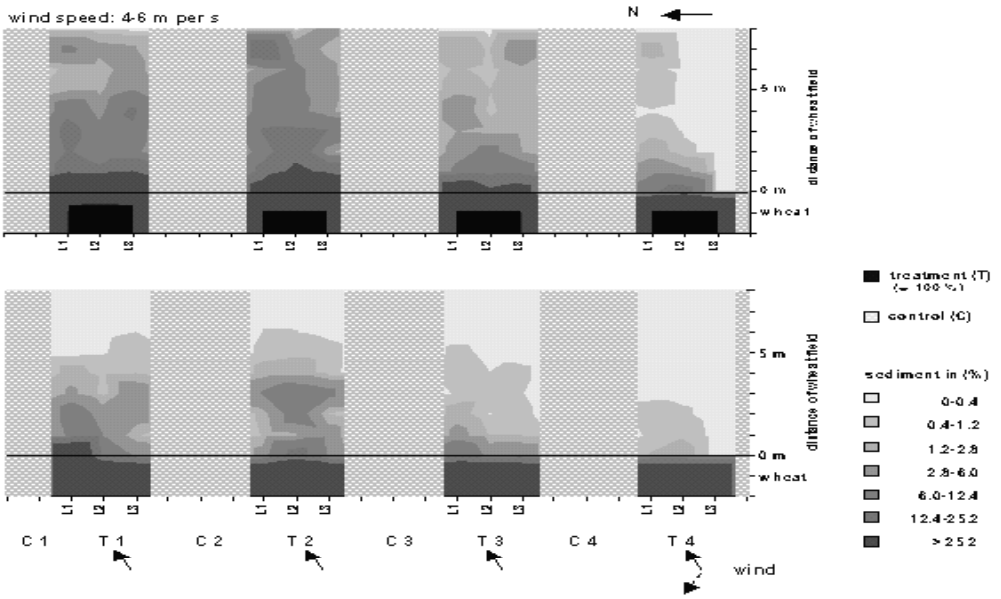

Figure 1 Sediment distribution on top of the vegetation (above) and on the soil surface (below) in the grass strip after lambda-cyhalothrin application in wheat, 1998, Osdorf

On the basis of the preliminary results, we found that the insecticide applications caused no significant effect (P=0.05) or only weak influence (tendency at P<0.05) on arthropods in field margins. The following Figures 2-4 demonstrate results yielded with different methods.

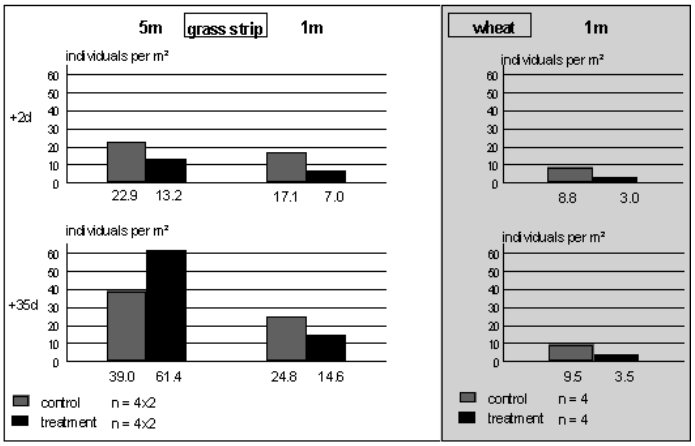

Figure 2 Results of biocoenometer surveys after lambda-cyhalothrin application, *Brachycera* community, 1998, Osdorf

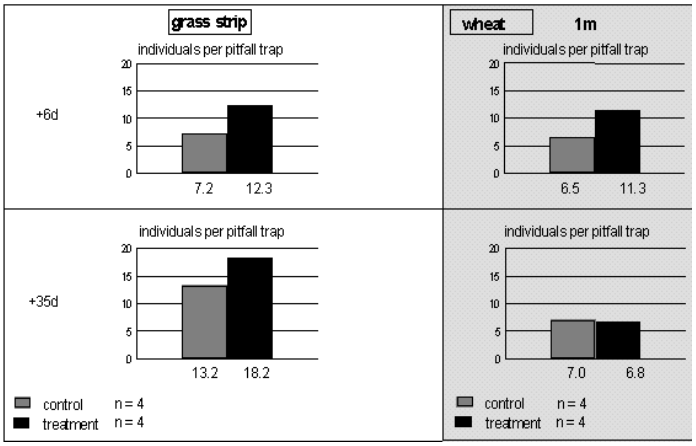

Figure 3 Results of pitfall trap captures after lambda-cyhalothrin application, carabid community, 1998, Osdorf

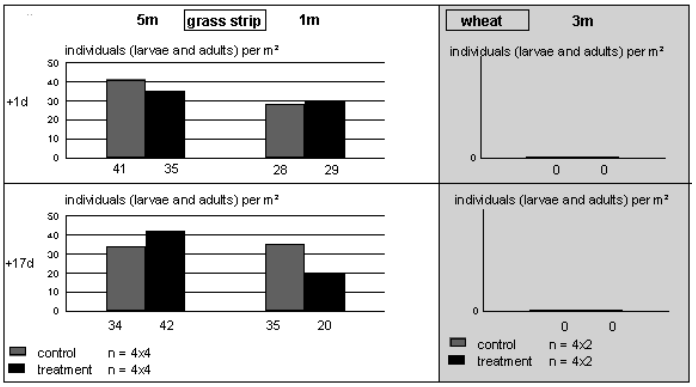

Figure 4 Results of grasshopper cage countings after lambda-cyhalothrin application, grasshopper community, 1999, Osdorf

Discussion

The preliminary experience and results have justified once again the general discussion on the best methodological approach to study accurately enough possible insecticide drift influences on field boundaries. Although time-consuming, it seems to be appropriate to use a combination of some different methods. From the statistical point of view the division of the investigated boundary into 8 plots gave the possibility of significance tests. But the variance and low level of densities per sample limited the statistical evaluation of the data (BROWN, 1998). A permanent problem seems to be the high demand to an accurate identification of the arthropods, especially because the ecologist must include a wide scale of different arthropods occurring in the samples. The response of all arthropods inhabiting the margins can not be determined. It is necessary to concentrate on suitable bioindicators to study real toxic effects and diversity changes as well as recovery processes.

In the present experiments, the distance-depending spray drift deposition was clearly identified by means of fluorescent marker. The deposition results confirm the findings by GANZELMEIER et al. (1995) and RAUTMANN et al. (1997) who investigated and defined base values for pesticide drift deposition on non-target areas. The concentration of the samplings on the 1 m and 5 m line beside the wheat crop allowed a special comparison of two distances to the sprayed field.

Results from the first investigation season led to the conclusion that side-effects on the non-target arthropod populations in the field margins seemed to be rather weak. However it must be considered that rain fell immediately after the application. This conclusion is supported by the preliminary results of residue analyses. However, the preliminary data from the following season 1999 also document only weak insecticide effects on non-target arthropods in the margins. A closing discussion of the results in detail is not possible before all determinations are finished.

Summary

The effects of insecticide Karate WG (7.5 g a. i. per ha, lambda-cyhalothrin) applied as 300 l (400 l in 1998) spray mixture in a wheat field on the arthropod community of two different adjacent boundaries is investigated over 3 years. The investigations started in 1998 and are focused on a 470 m long grass-dominated herb strip next to the field. To get true replicates, the strip has been divided into 8 plots (4 each as treatment and control). The following methods were used: biocoenometer sampling, Malaise trapping, pitfall trapping, grasshopper cage surveys, measurements of the contamination of field margins (fluorescent marker BSF), residue analyses (leaves, soil), studies on exposure of non-target arthropods (carabids, spiders). All the investigations were carried out along a 3 m line in wheat crop running parallel to the field border and along to lines through the field margin at 1 and 5 m distances to the field. Only the pitfall traps were placed in crop and margin along a 3 m distance line. Although the contamination measurements have shown clear drift indications till at least 6 m into the herb strip, significant effects could not be established on the basis of the results from 1998 and 1999 evaluated till now. However, there are some tendencies of certain effects on grasshoppers, spiders and carabids in the 1 and 3 m lines. The results are discussed in connection with methodological and statistical problems.

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The metapopulation concept and the use of GIS-tools for environmental risk assessment and management

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Abstract

There is now a general awareness amongst ecologists that many natural populations are most appropriately viewed not as single homogenous units, but as more complex structures, consisting of local semi-autonomous populations, linked to each other by dispersal. This "metapopulation" perspective is particularly important when one has a specific interest in estimating the impact of local perturbations on the longer-term dynamics of populations. In this paper, we present the results of an extensive mark-recapture study of odonates (dragonflies and damselflies) and show how this information can be readily incorporated into landscape-scale ecotoxicological models. By accounting for the observed dispersal characteristics of odonates within a Geographical Information System, we can not only estimate the rate of immigration of odonates into any pond, but also estimate their rate of recovery following a toxic perturbation.

Introduction

Over the past 50 years, increasing emphasis has been placed on the use of agro-chemicals which are of benefit to agriculture, but which have relatively few adverse effects on the ecosystems in which they are employed. One way to reduce the impact of a chemical is to vary the recommended dose, timing and extent of application, according to location. To a degree, farmers already take landscape features into account, for instance through reducing sprays close to field margins, but with the rapid development of tools such as Geographical Information Systems (GIS) there is the prospect of allowing managers to employ site-specific recommendations, rather than these general rules.

It is now widely recognised that in any area the total population of individuals of any one species are usually formed from smaller sub-populations that are linked to each other by dispersal (a “metapopulation”, HANSKI & GILPIN, 1991). As the rate of immgration of individuals into any particular sub-population will depend on the number, sizes and location of source populations, then the rate at which local populations recover from a serious perturbation is likely to be heavily dependent on the landscape in which the sub-population is found (SHERRATT & JEPSON, 1993). Whilst the theory is reasonably well understood, there remains a clear need for: more quantitative information on dispersal of species of relevance to ecotoxicological risk assessments and (b) modelling tools that allow us to use this information to estimate recovery in real landscapes (FAHRIG & FREEMARK 1997).

Plate 1 *Libellula quadrimaculata* L. bearing mark on forewing.

To address this shortfall, we have investigated the dispersal behaviour of odonates (damselflies and dragonflies) in agricultural landscapes and employed this information to parameterise a suite of spatially-explicit models of ecological recovery (CONRAD et al., *in press*, SHERRATT et al., 1999). Odonates were chosen for several reasons. First, they are generalist aquatic predators and as such they are particularly vulnerable to agricultural chemicals (as our data have incidentally demonstrated – see Results). Many are also of conservation interest (CORBET, 1980). Of particular relevance to this study, is the fact that they breed only in water. By focussing on those species which reproduce in discrete habitats such as ponds, then it is possible to identify all the potential sources and destinations of breeding adults using an appropriate GIS. Finally, odonates are highly visible, day-flying, easily handled and large enough to be caught and marked with an individual code painted onto a wing. Surprisingly, for such a highly visible and charismatic class of insect, empirical data on the characteristics and extent of odonate dispersal are seriously lacking (MICHIELS & DHONDT, 1991).

Methods

Mark-recapture experiment

Our study site, which was situated close to Churton, Cheshire, UK, covered an area of over 100 ha and included 11 marl ponds (615-1300 m^2 area, 30- 860m apart) (Fig. 1). All the ponds were situated in fields that were used for silage or pasture, except Bunting and Karen A,B & C ponds which were situated in potato fields. From 21st May – 28th July 1997, a total of 4618 odonates from 13 species were individually measured, sexed, aged (teneral or adult) and marked with unique code. Adults were captured for marking at Karen A, Swan, Bull & Leccy ponds between 8.00am and 10.00am each day (weather permitting). Another separate sample was caught and marked along hedgerows. The rest of the day was spent systematically visiting every pond in the study area 3 times a day for 20 minutes each time, in order to estimate odonate densities and search for marked individuals. When manpower permitted, searches for marked individuals were also conducted up to 2 km outside the study area. In mid June, an aphicide-fungicide mixture was applied to the fields that surrounded Karen A, B & C ponds. Although we were informed of this event, it was not part of our experiment. However, we continued to monitor odonate densities around these ponds to determine the effect of this pesticide on their dynamics.

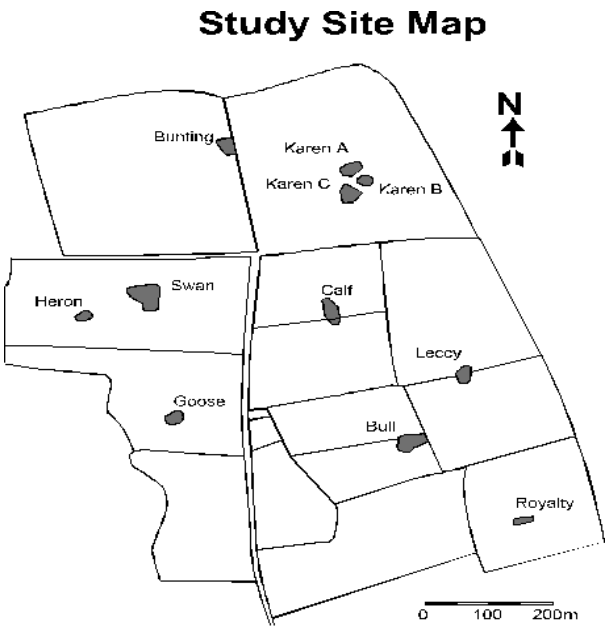


Figure 1 Study site near Churton, Cheshire UK (53°6' N, 2°51'W)

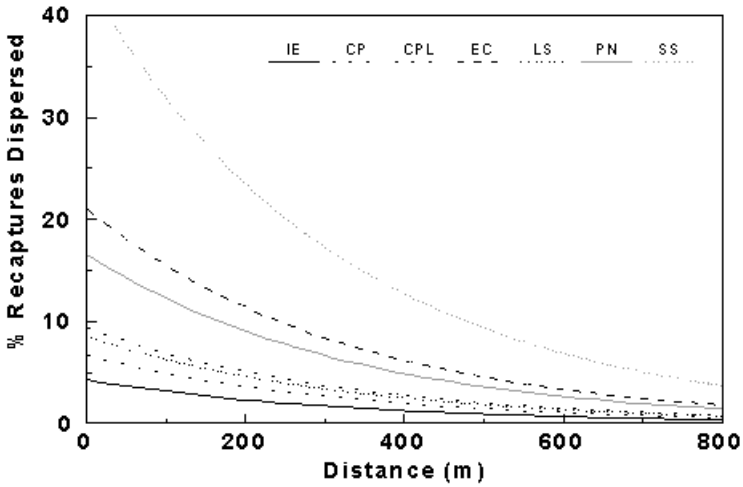
Modelling

In cooperation with Cheshire and Durham County Councils, we manually edited and extracted all water polygons and field boundary features within these two counties from Ordnance Survey Landline 1:2500 scale digital vector map data. These coverages were held in an Arc/Info GIS (ESRI, 1998). Since our field studies had indicated that very few damselflies disperse more than 2 km, we identified every pond in these two counties above $10m^2$, and calculated the sizes and proximities of all the ponds within 2km from each and every pond. By making the simplifying assumption that the number of damselflies of a given species around a given pond was directly proportional to the pond perimeter (in this case 10 individuals m^{-1} pond edge per season), it was possible to calculate the expected number of immigrants of any damselfly species per season into any pond in Cheshire or Durham. These coverages were intended to provide a useful first guide to the likely rate of colonisation by odonates, hence the recovery potential, of any pond of interest in these areas.

Results

Mark-recapture experiment

Overall there were 3085 re-sightings of 1631 marked individuals. From this data it was evident that the probability of individuals of all species moving between ponds declined exponentially with the Euclidean distance between ponds (Fig. 2). Systematic surveys of the whole area revealed that odonates had greater tendency to move along hedgerows than across fields, but the exponential model did not explain significantly greater variance when distance was expressed as ecological distance (minimum route between ponds along hedgerows). While all of the 7 species analysed showed this exponential relationship (Distance $F_{1,52} = 12.37$, p=0.001), which declined at a similar rate for all species ($F_{6,46} = 1.74$, $p = 0.13$), ANCOVA revealed that the overall tendency of individuals to disperse differed significantly between species ($F_{6,52} = 5.04$, $p < 0.001$). In general, dispersal probability appeared to increase with increasing species size, with larger species more likely to disperse further.



IE – *Ischnura elegans*; CP- *Coenagrion puella*; CPL – *C. pulchellum*; EC – *Enallagma cyathigerum*; LS – *Lestes sponsa*, PN – *Pyrrhosoma nymphula*; SS – *Sympetrum sanguinium*

Figure 2 Back transformed fitted regression lines from ANCOVA showing the % of recaptured individuals that dispersed versus distance dispersed. Only the 7 most commonly observed species were analysed.

Why did some individuals of a particular species stay at their natal pond while others dispersed?

The rarity of recording damselfly dispersal has usually prevented examination of whether some individuals are more likely to disperse than others (ANHOLT, 1990). To improve the statistical power of our analyses to detect differences we concentrated on two species for which we had the most data: the blue-tailed damselfly

(*I. elegans*) (2841 individuals marked) and the azure damselfly (*Coenagrion puella* L.) (769 individuals marked) and we divided our data set of recaptures into movers and non-movers. The features that characterise a disperser are currently being investigated in some detail (CONRAD et al., in prep), but for the purposes of development of the ecotoxicological models, it is important to note that there was no evidence that the tendency to move was influenced by the density of individuals at the pond. Furthermore, dispersers and non-dispersers could not be distinguished on the basis of their sex or age (teneral or adult) ($P > 0.05$).

The day after the fungicide-aphicide was applied to the potato fields, the number of odonates recorded around the ponds within these fields fell to zero (Fig. 3), but the recorded densities of odonates at other ponds not situated in the potato fields showed no apparent change. Whatever the route of exposure, it is evident that the insecticide temporarily reduced the size of the adult population around these ponds.

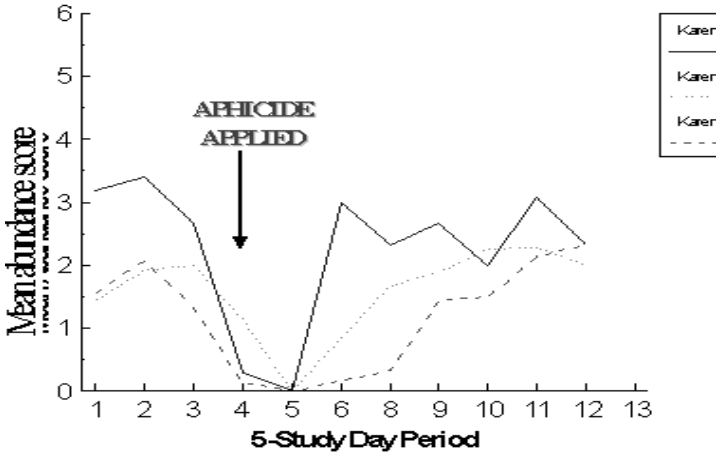


Figure 3 5-day average abundance scores of I. elegans recorded around the ponds Karen A, B, C before and after the application of a pesticide. 1= 1, 2=2-10,3=11-20,4=21-30

Modelling

The ultimate goal of our research program was to develop a tool which can map real populations and provide an assessment of their ability to recover from exposure to toxic chemicals. 15,935 ponds were identified in our Cheshire GIS, while County Durham contained only 448 recognisable ponds. The mean distance of ponds in Cheshire to their nearest neighbour was 139 m (sd 136.05), while the mean nearest neighbour distance of ponds in Durham was 378 (sd 409.23). Since our field studies had indicated that very few damselflies disperse more than 2km, we identified every pond in these two counties above $10m^2$, and calculated the sizes and proximities of all the ponds within 2km from each and every pond. By making the simplifying assumption that the number of damselflies of a given species around a given pond was directly proportional to the pond perimeter (ongoing work will allow us to refine this assumption, for instance by accounting for geology and land-use), it was possible to calculate the expected number of immigrants of any damselfly species per season into any pond in Cheshire or Durham (Fig. 4)

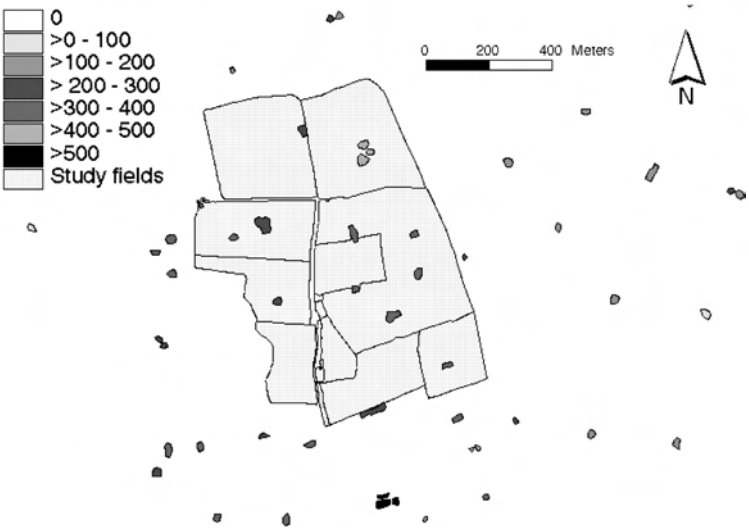


Figure 4 Estimated number of immigrants of *Coenagrion puella* per season into ponds around our Cheshire study site, enlarged from coverage of the whole county. Immigrants to each pond were estimated using dispersal data, distance between ponds and perimeter of the source ponds

Clearly this analysis is based on a suite of simplifying assumptions but for the first time an investigator can view the coverage of these counties and obtain a reasonable indication of the number of colonists a pond is likely to receive on the basis of its proximity to the network of other ponds in the area. Network and path analyses can be applied to these estimates, to highlight ponds which either act as sources of significant numbers of dispersers, or act as crucial "stepping-stones" linking otherwise poorly connected clusters of populations. In addition, by incorporating these estimates of immigration into appropriate stage-structured population models of damselflies and dragonflies (e.g., see CROWLEY et al., 1987), it is possible to obtain estimates for the time taken for odonates to recover from a given toxic perturbation. Such information is vital if we are to develop location-dependent guidelines on the application of toxicants.

Discussion

Our estimated rates of dispersal were somewhat higher than one might expect given the preliminary indications in the literature (e.g., see CORBET, 1980; FINCKE, 1982; BANKS & THOMPSON, 1985). However many of these earlier studies concentrated primarily on the behavioural ecology of damselflies and were not specifically designed to estimate dispersal rates. The fact that there were no consistent differences between movers and non movers, coupled with the fact that movement probability fitted a negative exponential function (typical of a simple diffusive process, BERG, 1983), suggests that dispersal of odonates was strongly influenced by chance events, for instance through individuals moving to a different pond after foraging in the hedgerows.

Although the pesticide treatment of the potato fields was unplanned, it serves to emphasise that odonate populations in agricultural landscapes may be vulnerable to xenobiotics. The fact that recovery was relatively rapid, and that newly-emerged tenerals were soon seen around the pond, suggests that the odonate nymphs in the pond were not directly affected. While due care appeared to be taken by the farmer with the pesticide application, it is possible that some of the pesticide drifted over the pond edges as it was being applied. Moreover, it is likely that odonates came into contact with the pesticide while foraging in the potato field.

There are few quantitative estimates of the rates of immigration of aquatic invertebrates to small water bodies, and since immigration will vary according to the size and number of local source populations, it is not surprising that these ecological characteristics are rarely considered in the risk assessment process. Our

GIS model is just one example of the way in which ecological and spatial information can be usefully combined. This model provides a simple tool which is capable of estimating the "colonisation potential" of any pond within Cheshire or County Durham, on the basis of its proximity to the network of ponds in the vicinity. We view this as an exciting development because it raises the real possibility of an ecologically-parameterised management tool, whereby the pattern and timing of application of agrochemicals in fields surrounding ponds can be planned to minimise adverse effects. Whilst the present study has concentrated on recolonisation of ponds by odonates, the approach is also promising for other groups of freshwater insect such as Trichoptera or Ephemeroptera, and may even be applied to consider organisms in other habitats, such as Carabid beetles or Linyphiid spiders in arable farming landscapes.

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Drift reduction by vegetation and application technique

Unsprayed field margins: a brief summary of the implications for environment, bio-diversity and agricultural practice

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Abstract

A management strategy has been developed for field margins to reduce pesticide drift to non-target areas and to promote bio-diversity on arable land. To this end, 3 and 6 m wide strips along the edges of winter wheat, sugar beet and potato crops have been left unsprayed with herbicides and insecticides and compared with sprayed edges. The effects on pesticide drift, arable flora, invertebrates, vertebrates, costs and acceptance to the farmer were studied. This article reviews the most important results.

Interviews with farmers show that field margins were sprayed intensively. Drift measurements demonstrated that the creation of unsprayed buffer zones of 3 m wide proves to be a very effective way of reducing pesticide drift to the ditch (by about 95%) and risks to aquatic organisms. In the unsprayed edges presence and abundance of arable plants increased substantially. The impact on epigeïc soil invertebrates was relatively minor. However, there was a pronounced effect on insects living on plants. The number of visits by *Motacilla flava flava*, an insectivorous bird, to the unsprayed edges was also higher. Cost-benefit analysis based on the yield losses shows that in winter wheat and potatoes unsprayed crop edges can well be adopted in agricultural practice. In sugar beet, however, the cost is too high. However, for agronomic, farming equipment related and socio-psychological reasons farmers prefer unsprayed cereal edges or grass strips in stead of potato edges. From their perspective a flexible width of the unsprayed crop edges is the most important aspect for acceptance in farming practice. The introduction of field margin management for environmental and bio-diversity aims is discussed.

Introduction

In recent years it has become clear that the natural values of arable land can be substantially enhanced by reducing pesticide use along field margins. This leads to a marked increase in the abundance of wildflowers, insects and birds in the unsprayed crop edges (see for example, BOATMAN et al., 1999). A major advantage of such crop edge management is that by implementing measures in a small portion of a field, it is in principle possible to guarantee species abundance over a much larger area. Moreover, this type of management also results in a reduction of pesticide emissions to the surrounding areas. Reduction of pesticide drift to surface water such as ditches is an important policy aim in a lot of European countries. In terms of agricultural management, crop edges are of less economic value than field interiors. The management of crop edges often requires additional effort and yields are generally lower. Together, these considerations indicate that a specific management regime for pesticide use along crop edges represents an instrument worth looking at more closely.

This article shortly reviews the results of a research project aimed at developing a management strategy for pesticide use in field margins in the Netherlands. More detailed information, including the complete reference list of project publications, is given in DE SNOO (1999). The study aimed to answer the following three key questions:

1. To what extent can a reduction in the use of pesticides on field margins decrease pesticide emissions to the field surroundings?
2. To what extent is it possible to enhance the bio-diversity in farming areas by decreasing pesticide use in field margins?
3. To what extent are the measures taken compatible with overall farm management?

Methods

Study area

The study took place during the period 1990-1994 and was carried out in the Haarlemmermeerpolder in the Netherlands. In this area the most common rotation on the farms is winter wheat followed by potatoes and a second winter wheat crop and finally sugar beet. To investigate the consequences for the environment, bio-diversity and the farm economy, strips 100 m long and 3 m wide were left unsprayed with herbicides and insecticides along the edges of these crops on six farms. In addition, for the bird research, in winter wheat, unsprayed strips measuring 450 x 6 m were created on 10 farms. The unsprayed strips were compared with sprayed strips, almost always in the same field. Spraying with fungicides was permitted and fertilizer regimes remained unchanged.

Sampling program

Environmental research

For the environmental research pesticide use along field margins was inventoried by means of interviews with 88 farmers. Next, measurements were made of the pesticide drift resulting from spraying of the field edge with knapsack sprayers and from crop treatment with field sprayers. The drift deposition occurring at various distances from the field was measured for different nozzle types and under various wind speed conditions using water-sensitive paper. A specific aim in the study with the field sprayers was to investigate to what extent creation of a 3 or 6-m wide buffer zone can reduce pesticide emissions to adjacent ditches and the risks to aquatic organisms. In the risk assessment the predicted environmental concentration in the ditch, of the pesticides most frequently used in the research area, was calculated on the basis of the drift deposition measured in the present study and compared with the toxicity for aquatic organisms.

Ecological research

In the ecological research the presence, abundance of the arable flora was studied. Relevés were also made on the ditch banks adjacent to the unsprayed and sprayed crop edges. Insects on the vegetation and epigeic soil invertebrates were collected using different techniques. Farmland songbirds were studied in winter wheat. Finally, a limited inventory of small mammals was carried out.

Socio-economic research

For the socio-economic research an economic cost-benefit analysis was performed on the fields, with the harvest losses (quantity and quality) in the unsprayed crop edges being compared with the savings on pesticide use. Next the social acceptance of the measures on the part of the arable farmers was investigated. Therefore 31 farmers, with experience with the management of unsprayed edges, were interviewed about their perception and preferences concerning field margin management (width of the margin, location in the field fixed or rotation, the payment system and the guidance). Besides unsprayed crop edges, the acceptance of other types of margins was also investigated, such as grass-sown margins and set-aside margins.

Environmental results

Pesticide use

It was found that field margins are sprayed very frequently with pesticides. The inventory showed that when spraying the crop, more than 85% of farmers also spray the field edge (a strip between the crop and the ditch bank of about 0.35 m wide). Moreover, 95% of those interviewed reported separate spraying of the field edge once or twice a year to create a sterile strip. In most cases glyphosate was used for this purpose. Almost 60% of farmers also spray ditch banks, generally with glyphosate and/or MCPA. In addition, another 18 compounds are employed on ditch banks, most of which are prohibited for this purpose. Finally, 30% of farmers also spray the ditch bed, when the ditch is (partly) dry in late summer (glyphosate and dalapon).

Pesticide spray drift

Pesticide drift of *knapsack sprayers* to the surrounding environment resulting from the use on field edges is found to be highly dependent on the wind speed and type of spray nozzle employed. At low wind speeds ($\leq$3 m/s) there is virtually no drift into the ditch: <0.1% of the deposition on the sprayed area (= 100%). On the ditch bank, on the other hand, there is up to 9% deposition (2.7% average). As the wind speed increases, so

too does deposition on the ditch bank and in the ditch itself. At a wind speed of 5 m/s the maximum deposition measured was 3.2% in the ditch and 30% in the ditch bank.

The measurements with field sprayers show that under practical working conditions pesticides are frequently deposited in ditches. The quantity deposited is highly dependent on wind speed. At hardly any wind (wind speed < 0.5 m/s) no deposition was measured in the ditch and a maximum of 6% halfway down the ditch bank. At a low wind speeds (≤3 m/s) these values were 2% and 25% respectively (average 1.1 and 16.9%).. At 5 m/s, a wind speed at which spraying still regularly occurs in practice, deposition in the ditch is approx. 7%. Risk analysis indicates that at this wind speed 12 of the 17 pesticides investigated pose a risk or even a high or very high risk to aquatic organisms. Creation of an unsprayed buffer zone only 3 m wide reduces spray drift to the ditch by at least 95%. With a buffer zone 6 m wide no deposition was observed in the ditch. The risk analysis indicates that with a 3-metre buffer strip only 4 of the 17 pesticides investigated pose a (minor) risk.

Ecological results

Arable flora

In all 3 crops the cover of arable plant species in the unsprayed edges was significant far higher than in the sprayed edges. In sugar beet cover increases from 10 to 44%, in potatoes from 4 to 11% and in winter wheat from 2 to 32%. On average, in sugar beet the number of species increased from 16 in the sprayed edges to 24 species in the unsprayed edges, in potatoes from 8 to 17 species and in winter wheat from 6 to 17 species. The increase was due mainly to an increase in dicotyledonous species. Many species were found almost exclusively in the unsprayed margins and nowhere else in the fields. Adjacent to the unsprayed winter wheat edges plant diversity and cover of dicotyledons increases also on the ditch banks.

Invertebrates

In the unsprayed winter wheat margins the insect density in the upper parts of the vegetation was 3 to 4 times higher than in the sprayed margins. This increase was due to flower-visiting insects such as Hover-flies (*Syrphidae*) and also to natural aphid predators such as Ladybirds (*Coccinellidae*). The number of insect groups also increased in the unsprayed margins, by a factor 1.4. Although aphids were more abundant in the unsprayed winter wheat margins, they did not spread to the rest of the field. The number of butterflies increased significantly in the unsprayed winter wheat and potato crop edges. The number of individuals found in the unsprayed winter wheat edges was 4 to 5 times higher than in the sprayed edges. In winter wheat, in both years investigated, and in potatoes, in one year, there was also a significant increase in the number of butterfly species in the unsprayed strip. On the ditch bank and field edge bordering on unsprayed crop edges, too, there was in one or both years a significant rise in the number of butterfly individuals and species.

Of the dominant ground-dwelling invertebrates, only the *Araneida* in winter wheat and the *Coleoptera* in sugar beet were trapped slightly more frequently in 1 of the 2 years of study in the unsprayed edges than in the sprayed edges (by a factor of approx. 1.2). With the *Carabidae*, constituting most of the bio-mass, the activity density in winter wheat (both years) and in sugar beet (one year) was significantly higher in the unsprayed margins. However, the increase in the number of insects trapped was relatively small. In these crops the number of species was significantly higher in one year. In potatoes there was only a significant difference in the number of species (in one year). At the species level, herbivorous *Carabidae* species show the most marked increase.

The bird censuses indicated that unsprayed winter wheat margins are very attractive to the Blue-headed Wagtail (*Motacilla flava flava*). Compared with the sprayed margins, these unsprayed strips are visited 3 to 4.5 times more frequently by this species. In the case of the Skylark (*Alauda arvensis*) no difference was found in the frequency of visits. The limited inventory of small mammals undertaken indicated more (field)mice visits in unsprayed than in sprayed cereal edges.

Dimensions of the margins for biodiversity

In the study unsprayed winter wheat margins were created of 3 and 6 m wide. It was thus possible to gain an indication of the margin dimensions that are most appropriate for promoting the abundance of flora and entomofauna. Comparison of the margins 3 and 6 m wide showed that especially the outer most metre of the

field that are important for the vegetation and the insects living there. Also in the sprayed situation the outer 3 m harbour the greatest number of species. In the unsprayed situation there is no extra increase in the abundance and presence of arable plant species or insects in the 6-m-wide strips relative to the 3-m-wide strips.

Socio-economic results

Cost-benefit analysis of unsprayed crop margins

In all the unsprayed margins the crop yield was lower than in the sprayed margins. On average over the various farms and years of study, the loss in yield amounted to about 30% in sugar beet, 2% in potatoes and 13% in winter wheat (11% in 6 m wide cereal edges). The considerable loss in sugar beet yield was due to the crop becoming overgrown with arable weeds. The harvest from the unsprayed margins was almost of the same quality as that from the sprayed margins. A comparison of the costs of harvest losses with the savings on pesticide use achieved in the unsprayed margins indicates that the net costs are high in sugar beet: approx. DG 0.10 per m^2. This type of field margin management can not be fit in agriculture practice. In a potato crop the costs of harvest losses are offset by the savings on pesticide use; with this crop there are zero nett costs,. in winter wheat, finally, the nett costs are low: approx. DG 0.005 per m^2.

Social acceptance of the measures

If several different management options are proposed to arable farmers, they express a clear preference for unsprayed cereal margins or unsprayed grass strips. Unsprayed margins in potato crops and set-aside strips score substantially lower. This choice is found to be based mainly on crop protection arguments, such as the increase in the abundance of arable plant species and in the occurrence of diseases and pests on the margins and in the rest of the field. In the case of grass-sown strips the lack of suitable farm machinery was quoted as the main argument. It is surprising that unsprayed potato margins score so low, despite the low costs they entail. This is probably due to a desire to avoid all risks in this high-profit crop. It was also found that the most important aspect determining the compatibility of field margin management with overall farm management is the width of the unsprayed margins. In this context there is a preference for a flexible width.

Conclusions

Summarising, it can be concluded that in the Netherlands field margin management offers promising perspectives. By creating a relatively narrow strip 3 m wide, current Dutch policy emission targets pertaining to pesticide spray drift to surface waters can be achieved. As a means of enhancing biodiversity in farming regions, the creation of unsprayed margins in winter wheat offers the most promising perspectives. The main effects appear to be an increase in the abundance and variety of arable plant species and their associated insect fauna. There appears to be relatively little impact on epigeïc soil invertebrates. The Blue-headed Wagtail also appears to benefit from the unsprayed edges. The different results for the two bird species are probably due to differences in diet and foraging strategy: The Blue-headed Wagtail forages in the upper parts of the vegetation, where there is a major impact on insect abundance, while the Skylark forages exclusively on the ground and is mainly herbivorous. In the unsprayed situation there is no extra increase in the abundance and presence of arable plant species or insects in the upper part of the vegetation in the 6 m-wide winter wheat strips relative to the 3 m-wide strips. This argues for the creation of long unsprayed margins 3 m-wide rather than 6-metre-wide strips of shorter length. By creating longer margins, moreover, there is greater benefit to this sort of bio-diversity and the environment along the ditch bank and in the ditch itself. In winter wheat the measures involve little extra nett expense. Finally, arable farmers have a positive attitude towards unsprayed cereal margins. In comparison with winter wheat, in potatoes the creation of unsprayed margins yields less conservation benefit. Although the costs in potatoes are extremely low on average, in this crop unsprayed margins are not popular with farmers. Also from the view point of environmental protection, the creation of unsprayed margins serves less purpose in this crop, because with many varieties of potato it is almost impossible to discontinue the use of fungicides. Through use of these chemicals aquatic organisms in adjacent ditches remain at risk. In sugar beet, finally, the costs of the measures are so high that they make unsprayed margins impracticable in this crop. In the case of potatoes and sugar beet a better option would be to create an unsprayed cereal margin or unsprayed grass-sown strip around the crop.

The introduction of a specific strategy for field margin management in the Netherlands may be feasible, as part of existing environmental policy. The basic principle is then: good agricultural practice means no

pesticides in the ditch. Environmental policy then has to determine the width of the buffer zone. From an environmental point of view an unsprayed field margin is just a tool, comparable with other drift reduction options, and not an aim in itself. (DE SNOO & CHANEY, 1999). Therefore, the answer to the question "Whether annually recurring costs in terms of yield reductions connected with the prevention of surface water pollution, should be reimbursed by government?", should be no. If farmers can tackle the pesticide problem in adjacent areas in an alternative way by creating unsprayed field margins, then it is their choice as entrepreneurs to make this decision.

From a bio-diversity point of view, two targets can be distinguished for the promotion of field margin management. The first is to create unsprayed field margins to stimulate rare or red list species. From a traditional conservation point of view this is a clear aim, especially in ecologically rich areas or to focus on endangered or specific species. To reach conservation targets it is often necessary to take rather drastic measures, such as reducing or omitting pesticides and fertilizers, which result in large crop yield depressions. Unsprayed field margins can be seen as refugia, where with sensitive management species can survive the direct (toxic) effects of pesticides. In many cases there is a specific responsibility of national governments or local authorities for the conservation of certain species laid down in specific policy. Due to the relatively high costs of these measures, farmers can be compensated by governmental organizations.

The alternative option for field margins, is not to focus on rare species, but to create unsprayed margins to compensate for the indirect effects of pesticides. This is a more ecological approach, with the philosophy that field margins play a role in the conservation of communities in arable fields. The basic conditions for wildlife on the farm - in terms of food and shelter - must be known before field margin management strategies can be devised. Depending on the ecological infrastructure on the farm and the type of landscape it is possible to determine which of the basic conditions are missing on the farm involved. If the quantity or quality of shelter is lacking, this can be enlarged by creating grass margins, other management etc. Where food is a problem, in most cases an unsprayed cereal edge is a good option. In general, the aim is for common species to be abundant in the unsprayed margins, because these form the basis of the food chain, and also to encourage rare species. In many cases it is not necessary to stop fertilizer use and the costs of these measures will be relatively low. It seems reasonable to argue that with this approach the government should not pay for the measures, because this should also be part of Good Agriculture Practice, and sustainable populations of non-target species should survive on arable land. To implement such field margin management on a broader scale it is necessary to have a step-wise approach based on a farm plan taking into account the existing habitats on the farm, the type of landscape involved and the criteria for the species on arable land to survive. This approach has been initiated on organic farms with the Farm-Nature Plan (SMEDING & JOENJE, 1999). This type of plans could be made part of a company certification, in which not only environmental aspects, but also criteria for nature conservation are covered (UDO DE HAES & DE SNOO, 1997). It should be possible for farmers to benefit from such a company certificate in terms of better marketing, because they can easily supply the retailers.

Acknowledgements

This paper is based on DE SNOO (1999) and DE SNOO & CHANEY (1999).

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Drift measurements in the Netherlands as a basis for differentiation of risk mitigation measures

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Abstract

A summary is given on field measurements of spray drift research for the past 10 years in the Netherlands. Results are presented for orchard spraying, nursery tree spraying and arable field spraying for the typical Dutch situation, related to defined distances and dimensions of the surface water. Spray drift research was set up in order to identify and quantify drift reducing technologies. Results are presented for cross-flow sprayers, tunnel sprayers, and air-assisted field sprayers. The effect of nozzle type on spray drift is highlighted both with a modelling approach as based on field drift experiments. The effect of spray drift reducing technologies in combination with crop- and spray-free buffer zones is outlined. It is concluded that the right choice of spray technology can be used to minimise spray- and crop-free buffer zones and maintain acceptable levels of ecotox in the surface water.

Introduction

The Multi Year Crop Protection Plan (MYCPP, 1991) of the Dutch government formulates objectives for a reduction in plant protection products to be used and for an application practice for these products which is safe and more compatible with the environment. The emissions of plant protection products to soil, (surface)water and air should be reduced. A general reduction in spray drift to surface water next to the sprayed field can be achieved by improvements in spray application techniques. For the last 10 years an intensive measuring programme on spray drift has been performed and summarised in this paper. Results are a basis for legislation dealing with the authorization of pesticides and the quality of the water.

In order to apply a risk assessment the results are presented on a uniform basis and expressed as percentage of the application rate per surface area, at a distance of 2,25-3,25 m (for field crops) or 4.5-5.5 m (for orchards) of the last crop row, being the place where commonly the ditches are situated (Fig. 1).

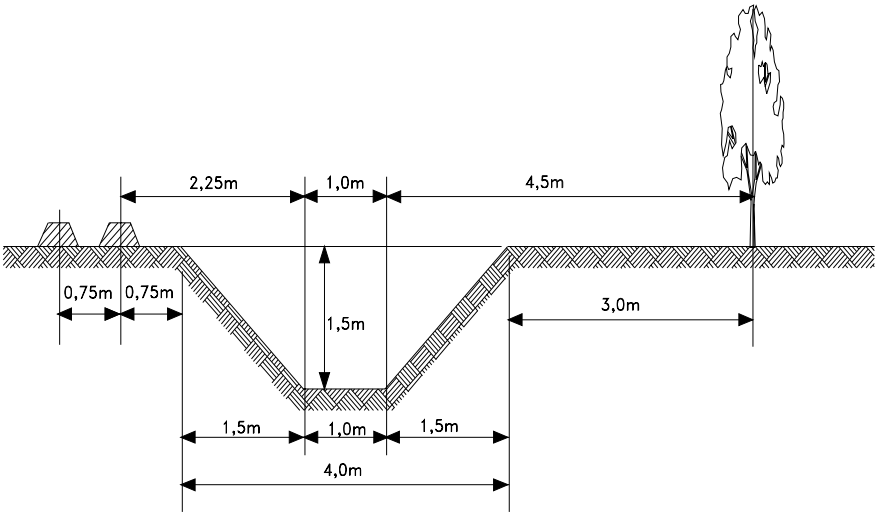

Figure 1 Representation of the place of the ditch, embankments and water surface, and the last rows of a potato crop and a tree row in an orchard (after HUIJSMANS et al., 1997)

When spraying for crop protection it is important that the chemical deposits on the right place and that coverage meets the needs for a good biological efficacy. Spray drift to zones adjacent to the sprayed field must be prevented as much as possible. The research programme consists of laboratory measurements, field experiments and computer modelling. A system analysis approach was developed to divide the research in spray processes and parts important for spray drift such as:

- Sprayer: nozzle (drop size, spray quality, driftability),
- sprayer boom movement and boom height (drop trajectory),
- sprayer outline and additional drift reducing technology,
- Crop: height, density,

placement of the last nozzle to the edge of the crop,

field layout and distance to the surface water.

The programme started with the quantification of the drift for the reference situation of the MYCPP, in which the drift level was set to e.g. 2% for field sprayers (based on an expert judgement). Then a stepwise approach was chosen to lower drift with: air assistance or shielding sprayer booms on a field sprayer, a tunnel sprayer, sprayer boom height and nozzle type. Different aspects will be highlighted in this paper, both for orchard spraying, nursery tree spraying as for arable field spraying.

Materials and Methods

Modelling

Spray quality and driftability are two important nozzle parameters in this context. Spray quality depends on nozzle type, nozzle size and spray pressure and is of importance for crop coverage. Drop size, drop speed, and drop direction in the spray fan influences driftability. Through a combination of laboratory measurements and computer modelling a driftability classification system is developed. With a PDPA-laser (Aerometrics; Phase Doppler Particle Analyser), spray quality and drop speed are measured. These data are used as input for the IDEFICS spray drift model (HOLTERMAN et al., 1997), calculating spray drift deposits downwind of the sprayed field. Spray drift is calculated for the zone 2.125-3.125 m from the last nozzle. In most cases this is the surface water area of the ditches adjacent to a potato field.

Field measurements

The developed methodology to classify spray nozzles for driftability holds only for conventional use of nozzles. Extension of the classification of driftability of nozzle types in combination with air-assistance, shielding, etc. on field sprayers still needs field measurements of spray drift.

In a series of field experiments air-assisted spraying was compared with conventional spraying in a potato crop during the growing season. The effect of low-drift nozzles on spray drift was also quantified, as well as the effect of a no-spray buffer zone. Measurements were done on a bare soil surface and in a ditch, downwind of the crop.

The measurements of spray drift are carried out according to the ISO-draft standard (ISOCD 12057; ISO/TC23/SC6N283 dated 01-08-1997) adapted for the typical situation in the Netherlands (ground deposits, ditch, surface water adjacent to the sprayed field). Measurements were carried out by adding the fluorescent dye Brilliant Sulfo Flavine (BSF) to the spray agent and placing collectors in and outside the field. The swath-width sprayed was at least 18 m. The length of the sprayed track was at least 50 m. A minimum of ten replications were made in time and place spraying along the edge of the field during the growing season, in order to meet average crop and spray conditions. The distance of the last downwind nozzle to the edge of the field (the last crop leaves) was determined. Measurements of spray drift were always compared to a reference situation, e.g. field sprayers applying a volume rate of 300 l/ha with a Medium spray quality. In case of air assistance, nozzles were kept vertical and air velosity was set to the maximum capacity of the fan.

Ground deposit was measured on horizontal collection surfaces placed at ground level in a double row downwind of the sprayed swath. When measuring field sprayers the collectors were placed at distances 0,5, 1-1,5, 1,5-2, 2-3, 3-4, 4-5, 5-6, 7,5-8,5, 10-11, 15-16 m from the last downwind nozzle. Collectors used were synthetic cloths with dimensions of 0,50 x 0,08 and 1,00 x 0,08 m.

Airborne spray drift was measured at a distance of 5,5 m from the last downwind nozzle. The collection of airborne spray was done on two seperate lines with attached collectors at 0, 1, 2, 3, and 4 m height. Collectors used were spherical synthetic cleaning pads (diameter 0,08 m) (no data presented).

After spraying, the dye was extracted from the collectors. The rate was measured by fluorimetry and expressed per surface area of the collector. The spray drift was expressed as percentages of the application rate of the sprayer (spray dose).

Meteorological conditions during spray drift measurements were recorded. Wind speed and temperature were recorded at 5 s interval at 0,5 and 2,0 m height, using cup anemometers and Pt100 sensors. Relative humidity was measured at 0,5 m height and wind direction at 2,0 m height.

Statistical analysis of the data was done using analysis of variance (ANOVA 5% probability).

Results

Modelling

The spray drift model IDEFICS is preliminary used as an evaluation tool for parameter settings on a conventional sprayer. The effect of nozzle selection, sprayer boom height, crop height, wind speed, etc. can be calculated to optimise settings for field tests or scenario studies on e.g. the effect of spray free and crop free bufferzones (VAN DE ZANDE et al., 1995).

Nozzles are classified into drift-reduction classes compared to a reference nozzle (BCPC Fine/Medium; SOUTHCOMBE et al., 1997) in a reference situation. Calculations are performed at a wind speed of 3 m/s, a crop height of 50 cm and a sprayer boom height of 50 cm above crop canopy. Nozzle-pressure combinations are classified accordingly. It was shown that the combination of nozzle type, nozzle size and spray pressure (Table 1) defines the spray drift (PORSKAMP et al., 1999).

Table 1 Classification of nozzle-pressure combinations for spray quality and driftability. Spray quality is classified according to BCPC. Spray drift reduction is quantified with the threshold nozzle Fine/Medium (Lurmark 31-03-F110 @ 3 bar) as a reference

Manufacturer	Nozzle type	Pressure [bar]	Spray quality	Driftreduction class
Delavan	LF-110-01	4,5	very fine / Fine	-90
Lurmark	31-03-F110	3,0	Fine / Middle	0
Lechler	LU 120-06S	2,0	Middle / coarse	50
Teejet	8008 VS	2,5	coarse/ very coarse	75
Teejet	8015 SS	2,0	very coarse / Extra coarse	90
Albuz	ADE3 oranje	1,5	Coarse	75
Albuz	ADE3 oranje	3,0	Middle	50
Albuz	ADE3 oranje	5,0	Middle	25
Lechler	ID 120-02	3,0	Extra coarse	75
Lechler	ID 120-02	5,0	Very coarse	75
Lechler	ID 120-02	7,0	Coarse	50
Teejet	TT11004	1,5	Very coarse	75
Teejet	TT11004	3,0	Coarse	50
Teejet	TT11004	5,0	Middle	-25
Teejet	DG11002	3,0	Middle	25
Teejet	DG11004	3,0	Coarse	50
Teejet	XR11002	3,0	Fine	-90
Teejet	XR11004	3,0	Middle	0
Teejet	XR11008	3,0	Coarse	50

Field experiments

MYCPP reference situation

The reference situation for the MYCPP for field crop spraying was a conventional field boom sprayer spraying a potato crop during the growing season with an average windspeed of 3 m/s. Crop height was on average 0.5 m above soil-surface and sprayer boom-height was 0.7m above crop height. Spray volume was 300 l/ha, spraying was done with a flat fan nozzle-type (BCPC-class Medium).

From field experiments performed in the period 1991-1993 (34 repetitions) it was found that the spray-drift deposition at the soil at 2.25-3.25 m downwind of the last row was 5.4% of the application rate per surface area (PORSKAMP et al., 1995).

Effect of spray volume and air assistance

In order to quantify the effect of spray volume and air assistance on spray drift, a number of drift measurements were executed in the period 1992-1994 (PORSKAMP et al., 1995). Spray volumes compared were 150 l/ha and 300 l/ha, resp. a Fine and a Medium spray quality (SOUTHCOMBE et al., 1997) and sprayer boom height was set to 0.7 m above the canopy of the potato crop. Within this volume range the Medium or Fine droplet size (resp. 52 and 34 repetitions) not significantly affected the drift deposition in the experiments. Spray drift deposition on the distance 2.125-3.125m from the nozzle was on average 5.3% for both nozzle types sprayed conventionally.

Compared to this reference situation (86 repetitions) a field boom sprayer with air assistance (70 repetitions) achieved a 50% reduction in spray drift on the soil surface at the same downwind distance.

Effect of crop free buffer zone

Increasing the distance from the crop boundary and therefor the last nozzle to the surface water zone by means of a non-cropped spray-free zone of 2.25 m (3 potato ridges) reduced the deposition by 70% on the strip of 2.125-3.125m from the field border (PORSKAMP et al., 1995).

Effect of shielding and air assistance

In a series of experiments in a flower-bulb crop (1993-1996) the drift deposition on the soil next to the sprayed field was measured (33 repetitions). The effects of air-assistance and a shielded sprayer-boom on a field-sprayer and a prototype tunnelsprayer for bed-grown crops were evaluated (PORSKAMP et al., 1997). Sprayers were equipped with flat fan nozzles, either a XR11003 or a XR11004 sprayed at 3 bar pressure. Sprayer boom height was set to 0.5m above a crop canopy of on average 0.3m. The field experiments were performed in tulips, lillys or a flower-bulb look-alike crop, cut mustard. No effect of these crop types was found on spray drift data. Also no effect was found of the used nozzle types on spray drift. A shielded sprayer boom and air assistance reduced spray drift deposition at 2-3m distance from the last nozzle with 50%. A tunnelsprayer for bed-grown crops (e.g., flower bulbs) reduced spray drift with 90%.

Effect of nozzle type and air assistance

In 1997 field tests on spray drift have been performed to quantify the effect of a ”low-drift” nozzle type and air assistance (MICHIELSEN & VAN DE ZANDE, 1998). A comparison has been made with use of a Hardi Twin sprayer using air assistance, and as a conventional sprayer without air. Nozzle types compared were a standard flat fan nozzle XR11004 sprayed at 3 bar pressure applicating 300 l/ha (36 repetitions) and a TT11004 sprayed at 1,5 bar pressure applying 200 l/ha (26 repetitions) at the same driving speed. Sprayer boom height was set to 0.5 m above crop canopy of a potato crop 0.5m in height.

Spray drift deposit on the soil surface was reduced by the use of a TT11004 by up to 60% at a distance of 2-3 m downwind. The effect of air-assistance as performed in this test (full air, nozzles kept vertical) is both for the standard flat fan nozzle (XR11004) as for the anvil nozzle type (TT11004) the same. Air assistance reduces spray drift in both cases with 70% on a distance of 2-3 m downwind.

In 1998 spray drift was quantified for a series of low-drift nozzle types all applicating a spray volume of 300 l/ha. With identical travelling speed, sprayer boom height (0,5 m above crop canopy) and liquid pressure (3 bar) the nozzle types: standard flat fan (XR11004), drift guard (DG11004), anvil flatfan (TT11004) and two types of injection nozzles (ID12004 and XLTD11004) were evaluated in the field (Michielsen et al. 1999).

All nozzles were used in a conventional way and with the use of air assistance (Hardi Twin, full capacity - nozzles kept vertical). Canopy height of the potato crop was 0,5 m. Results show that the terminology low drift nozzle needs further specification because within the group of low drift nozzles a ranking towards level of drift reduction is possible. Compared with the XR11004 nozzle, as a reference situation for most spraying applications, it shows that e.g. for the 300 l/ha the ranking for drift reduction evaluated as soil deposit at 2-3m distance from the last nozzle is: 57% for the TT11004, 76% for the DG11004, 87% for the ID12004 and 88% for the XLTD. In combination with air assistance this ranking was: 82% for the XR11004, 89% for the DG11004, 90% for the TT11004, 96% for the ID12004 (Fig. 2) and 96% for the XLTD. The reduction of spray drift because of the use of air assistance seems to be independent of the nozzle type around 70%.

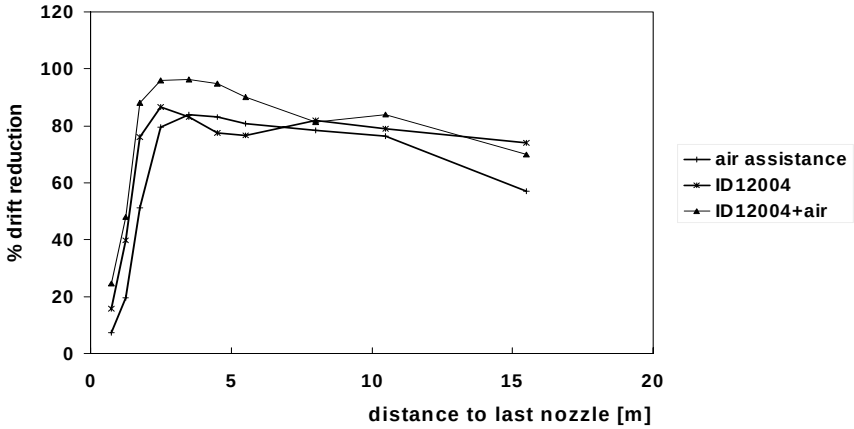

Figure 2 Spray drift reduction because of air assistance and nozzle type selection on different distances next to a potato field relative to a conventional spraying (XR11004 @3bar, 300l/ha)

Effect sprayer boom height

A comparison of the results from the experiments in the period 1992-1994 and 1997-1998 indicates a positive effect of reducing sprayer boom height above crop canopy on spray drift reduction, when spraying at 300 l/ha. Although not measured in the same experiment, based on the number of replicates, the conclusion can be drawn that a decrease in sprayer boom height from 0.7 to 0.5m above a 0.5 m crop canopy reduces spray drift with 70% on the distance 2-3m from the last nozzle when spraying a potato crop. When sprayer boom height is reduced the effect of air assistance on drift reduction increases. Where drift reduction is on average 50% for the 0.7 m boom height it increases to 70% for the 0.5 m boom height.

End nozzle

The effect of overspray of plant protection products when spraying the edge of the field can be reduced by the use of an end-nozzle (VAN DE ZANDE et al., 1995). An end nozzle produces a cut-off spray fan like from an off center (OC) or UB nozzle type. Depending on the placement of the last nozzle towards the crop-edge the nozzle is placed in the last nozzle connector or 0.2 m more to the outside (potatoes) (Figure 3).

Spray drift reduction, when using an end nozzle (UB8504) in combination with a low drift nozzle (DG11004) was around 20% on 2-3 m distance from the last nozzle (MICHIELSEN et al., 1999). On 1-2 m distance this effect is 50%. When using air assistance the drift reduction is resp. 60 and 80%.

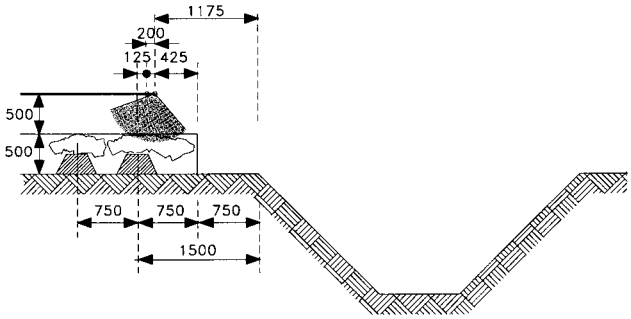

Figure 3 Placement of an end nozzle in a typical situation for spraying potatoes

Orchards

The reference situation for orchard spraying (Figure 4; top) is a cross-flow fan sprayer spraying in an orchard at full leaf stage (LAI 1.5-2) and an average windspeed of 3 m/s. The spray-drift deposition on the soil at 4.5-5.5m downwind of the last tree is 6.8% of the application rate per surface area.

Compared to this reference situation a tunnel sprayer (Figure 4; middle) achieves a reduction in spray drift on the soil surface of 85% and a cross-flow fan sprayer with reflection shields of 55% (HUIJSMANS et al., 1993). Spraying trees without leaves increases spray drift 2 to 3 times compared to spraying trees with full foliage.

A wind-break on the outer-edge of the field (Figure 4; bottom) reduces spray-drift 70-90% on the zone 0-3 m downwind of the wind-break (PORSKAMP et al., 1994).

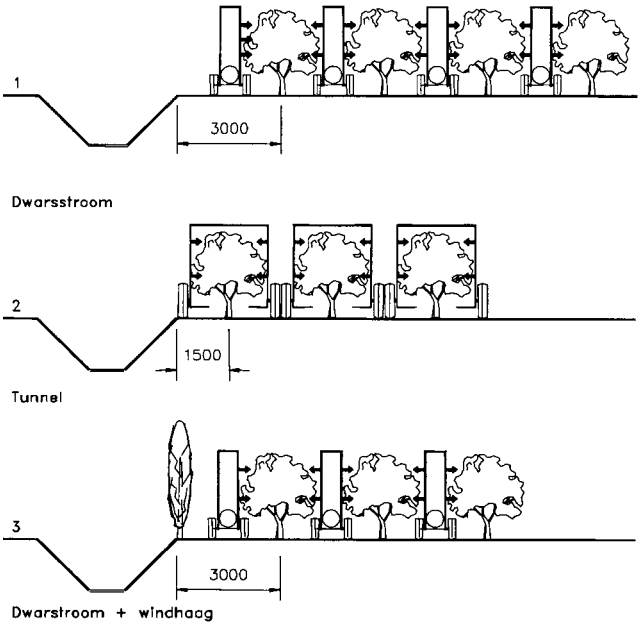

Figure 4 Representation of used spraying systems and situations in orchard spraying. Top 1: Cross-flow sprayer spraying last tree row towards the field; Middle 2: Tunnel sprayer; Bottom 3: Cross-flow sprayer with a hedge-row planted on the edge of the field

Nursery trees

In nursery tree growing a distinction is made between small crops (ornamentals) and large crops (lane trees), sprayed resp. downward and side- and upward.

In small grown crops spraying is usually done with a small hand-held sprayer boom, using a spray boom-height of 30 cm above the crop. Spray drift soil deposit on water surface was quantified as 1.6% of the application rate per surface area (VAN KAMMEN et al., 1998). A wind-break shield (50% open) placed on the edge of the field reduced spray drift with 60-80% compared with a conventional hand-held sprayer-boom spraying.

In a series of experiments (1996-1997) in lane trees, an experimental cross-flow sprayer and a conventional sprayer equipped with flat-fan nozzles were compared with a conventional axial fan sprayer with hollow cone nozzles (PORSKAMP et al., 1999). The comparison (16 repetitions) was made for two tree types: spindle form and transplanted alley-trees.

The level of spray drift deposition next to the sprayed field differs for the two tree types. When spraying with a conventional sprayer, the spray drift deposition on the soil at 3-4 m from the last tree row was, for the transplanted trees (13,6%) and for the spindle trees (3,3%). For both tree types no difference in spray drift was found for the conventional sprayer with flat fan nozzles.

When applying pesticides with the experimental sprayer, a significant drift reduction was achieved, exept for the spindle trees in the third growing season. Drift reduction averaged 50% (38-79%) compared to the conventional sprayer, whereas for spindle trees 165% more spray drift was found in the third season (pruned stems).

Discussion

Results from IMAG spray drift research (HUIJSMANS, 1997) are incorporated in Dutch legislation. In the Surface Water Pollution Act (VWS/VROM/LNV, 1999) and the Pesticide Act (VROM/LNV, 1998) criteria for drift deposit on surface water are used depending on spraying technique, crop free buffer zone and period of use during the growing season.

In the Water Pollution Act, packages of drift measurements are described to be implemented on the outside 14m of the fields by Dutch farmers. For the sectors arable farming, nursery tree or fruit growing minimal spray- and crop free buffer zones are described depending on the used spray drift reducing measures (Table 2). A minimum drift reducing package for arable farming is the use of low drift nozzles, a sprayer boom height of 0,5m and an end-nozzle (see Fig. 2). A low drift nozzle is defined as a nozzle reducing drift at least 50% in comparison with the Fine/Medium threshold nozzle from the BCPC nozzle classification scheme (SOUTHCOMBE et al., 1997).

Table 2 Spray and crop-free buffer zones in arable farming for different combinations of drift reducing actions (from VWS/VROM/LNV, 1999)

Action	Spray and crop free buffer zone in m
No drift reducing measures	14
Minimal drift reducing action	1,5
Minimal drift reducing action	
+ Catch crop on field boundary	1,0
+ Air assistance	1,0
Tunnel sprayer for bed grown crops	1,0

Note: The spray drift deposition level in these cases is set to the 1% level, which is in accordance with the results from field experiments in potatoes (MICHIELSEN et al. 1999)

A historical overview of what has been achieved in common agricultural practice over the last 5 years is presented in Table 3. Up till 1995, as quantified for the MYCPP, agricultural practice resulted (sprayer boom height 0,7 m) in a spray drift of 5.4% on the surface water distance 2.125-3.125 m from the last nozzle when

spraying potatoes. Good agricultural practice stated that sprayer boom height was 0.5m above crop canopy. In doing so spray drift was reduced to the 2.9% level. With the new incentive of the Water Pollution Act the use of low drift nozzles and an end nozzle is obligatory on the outside 14 m of the field. In combination with a crop free zone of 1.5 m spray drift deposition is reduced to 0.9% The use of a venturi nozzle instead of the minimal advised low drift nozzle reduces spray drift down to a level of 0.7%. The use of air assistance reduces spray drift in all situations with 50% (sprayer boom height 0.7 m) to 70% (sprayer boom height 0.7 m), independent of the used nozzle type.

Table 3 Spray drift deposition on water surface distance for potato growing in the Netherlands for the situations 1995, 1998 and 2000 depending on spraying technique and crop-free buffer zone

Situation	Crop free buffer zone (m)	Year of tests	Nozzle type	Sprayer boom height (m)	Air-assistance	Drift deposition (%)
1995	0,75	‘92-’94	4110-18	0,70	No	5,4
1995	0,75	‘92-’94	4110-18	0,70	Yes	2,7
1998	0,75	‘97+’98	XR11004	0,50	No	2,9
1998	0,75	‘97+’98	XR11004	0,50	Yes	0,6
2000	1,50	1998	DG11004 + end	0,50	No	0,9
2000	1,00	1998	DG11004 + end	0,50	Yes	0,15
2000	1.50	1998	ID12004	0.50	No	0.7
2000	1.00	1998	ID12004	0.50	Yes	0.15

The outlined spray drift reduction measures are in many cases overruled by the ecotox values of plant protection products to be met. Going down to levels lower than 0.2% spray drift is not exceptional and needs therefor further research on this subject. This holds also for the basic reason for spraying: crop protection with ensured biological efficacy. As in many cases spray drift reducing measures are not evaluated for its biological results with pesticides.

The results demonstrate that based on spray drift research a differentitated pesticide and water quality policy can be outlined and performed. The right choice of spray technology can be used to minimise spray- and crop free buffer zones and maintain acceptable levels of ecotox in the surface water. Spray technology plays a key role in the environmental risk assessment of pesticides.

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Drift reduction by vegetation

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Introduction

New environmental legislation has led some pesticide regulatory authorities in Europe to promote the use of approved spraying equipment and special practices to reduce spray drift (e.g., DE Official list of drift reducing techniques, Buffer-zones, UK Local Environmental Risk Assessment for Pesticides LERAPs). The operational codes of practice that have been developed make the underlying assumption that environmental exposure to spray drift can be minimised by adjusting the output of the sprayer to suit the target crop. Furthermore, this process of minimisation is limited by the pesticide formulation and sprayer design.

WALKLATE et al. (2000b) have shown that a high proportion of the variability of spray distribution, including off-target contamination, can be related to crop structure and atmospheric conditions, leaving aside the influences of pesticide formulation and choice of sprayer. However, the models that have been developed to quantify spray drift and identify situations that will reduce environmental contamination have tended to neglect important effects of the interactions between spray droplets and the surrounding vegetation.

This paper gives a summary of recent research at Silsoe Research Institute and Horticulture Research International at East Malling aimed at establishing the level of drift reduction that can be expected when key properties of orchard vegetation are taken into consideration and appropriate sprayer adjustments are made. The structural conditions of vegetation that give significant reductions of environmental exposure to spray drift have been identified. Drift reductions are estimated using a dose adjustment calculation based on the tree area density model of WALKLATE et al. (2000a). Finally, it is shown that windbreaks, commonly used in the UK to minimise wind damage, also yield a significant reduction of drift contamination outside the orchard.

Materials and methods

The following description gives an outline of the methods that were used establish the results that are discussed in the next section. For further details of the experimental materials and methods the reader is referred to the following reports: WALKLATE (1998); CROSS et al. (2000) and RICHARDSON et al. (2000).

Spray tracer distribution measurements - Orchards

Three different types of apple orchard were selected to represent the range of structures used for Cox apple production in the UK (i.e. established semi-dwarf trees on MM106 rootstock, established dwarf trees on M9 rootstock and 2-4 year old dwarf trees on M9 rootstock). Experiments were carried out in these orchards over a period of three growing seasons to measure the spray distribution deposit on the leaves of target trees and the off-target deposit on 2 mm diameter line collectors. A diagram of the layout of these experiments is shown in Figure 1. EDTA chelates of different metal salts were used as aqueous spray tracers to enable multiple exposure of each orchard to different treatment. The treatments were applied by using a typical axial fan sprayer (Figure 2 – Top left picture) set-up to give a range of perturbations of a standard set of operating conditions: spray flow rate of 11 l min^{-1}, sprayer speed of 1.6 m s^{-1}, air volumetric flow rate of 11 m^3 s^{-1} and a "very fine" spray quality with the following cumulative volume distribution diameter characteristics: D_{90}=244, D_{50}=156, D_{10}=87 µm. The different spray tracers were recovered from leaves on the target trees and other off-target collectors by using a standard aqueous wash-off technique. The tracer deposit samples were then measured by atomic absorption spectroscopy. Typical values of the mean distribution of spray are shown in Figure 3.

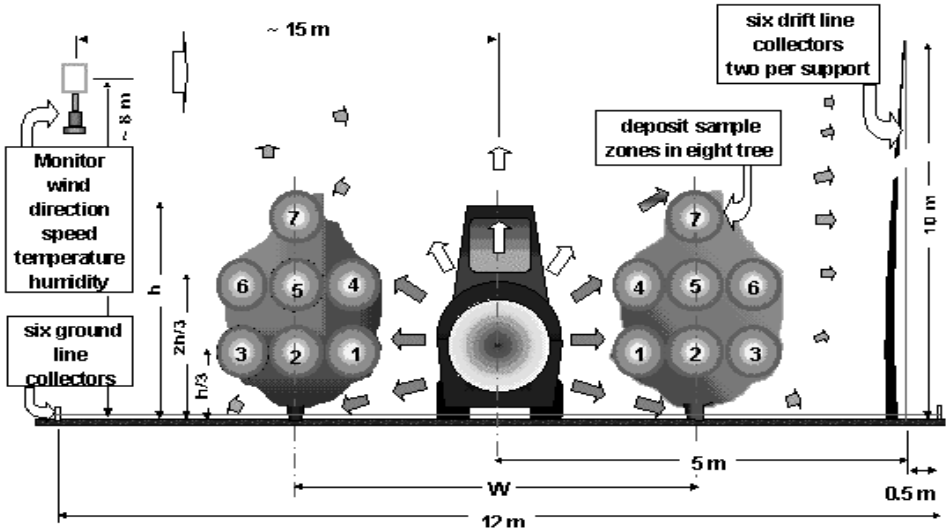

Figure 1 The layout of field equipment used to measure the deposit distribution on the leaves of target trees and off-target deposit distributions on 2 mm diameter line collectors

Spray tracer distribution measurements - Windbreaks

A limited number of distribution measurements have been made around a well-maintained windbreak of 7 m alder trees shown in the background of Figure 2. Drift measurements were made at different times throughout the growing season using 10 m vertical line collectors placed 2 m upwind and 2 m downwind of the windbreak during exposure of a single avenue of trees in an adjacent orchard. The analysis of spray tracers from these collectors was identical to that used for the orchard spray distribution measurements. Typical measurements of the mean distribution of spray are shown in Figure 6.

Meteorological measurements

Wind speed and direction together with relative humidity and temperature were measured at a height of approximately 8 m above the ground at a distance of 15 m upwind of the sprayer. Figure 2, left hand picture, shows this equipment mounted on a mobile mast so that local measurements can be easily made at different orchards.

Crop structure measurements

Tree and windbreak structures were recorded by using a tractor mounted LIDAR system shown in Figure 2.

Approximately 640.000 range measurements, accumulated during each 40 m traverse of the orchard, have been used to determine the two-dimensional interception probability distribution of each crop ”seen” by the LIDAR. WALKLATE et al. (2000a) have described the calculation of the crop parameters that have been used to provide scaling rules for matching the spray output to target deposit (i.e. Tree height, KOCH, 1993), Tree row volume (SUTTON & UNRATH, 1984) and Tree Area Density (TAD). Furthermore, it was shown that TAD gave better correlation with leaf deposit and off-target contamination than the alternative parameters.

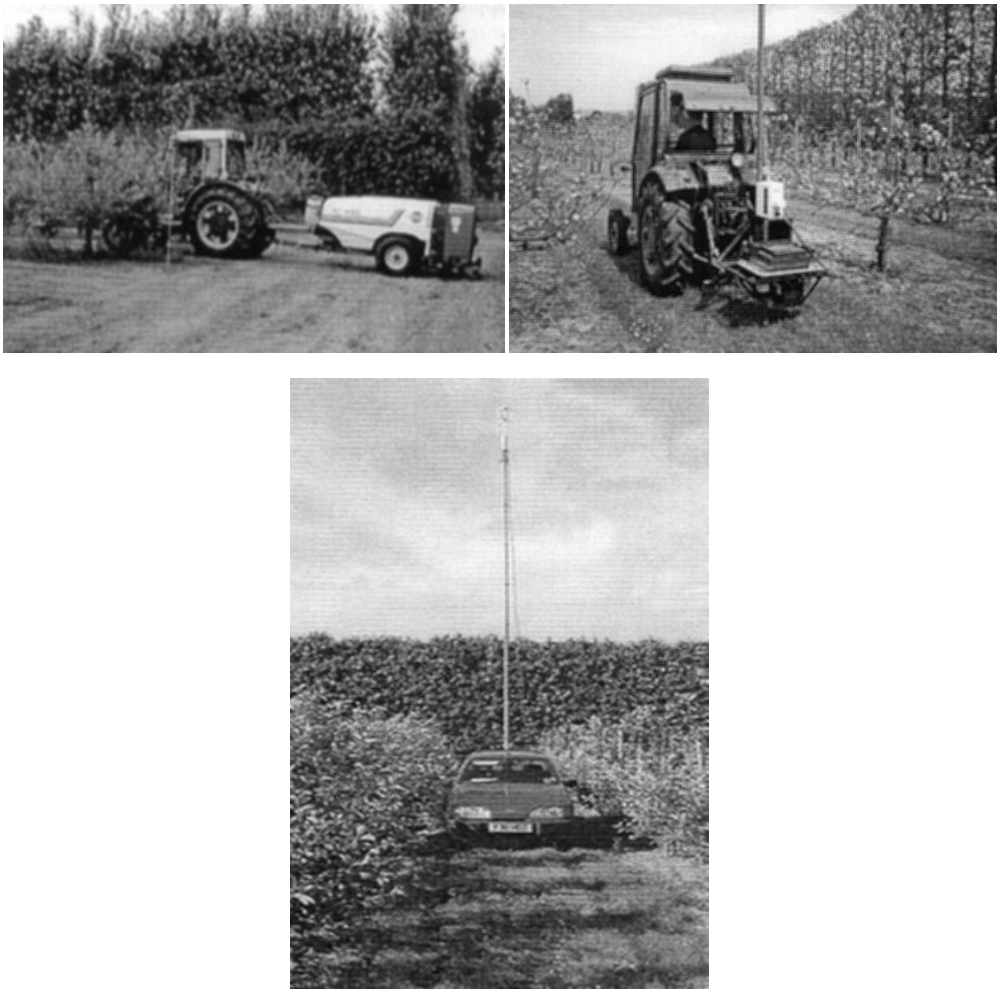

Figure 2 Photographs of equipment. Top left – The axial fan orchard sprayer. Bottom – The mobile equipment used to monitor the local wind conditions during exposure of different orchards. Top right – The tractor mounted LIDAR system used to measure structural parameter of orchard trees. The photographs also show various views of typical apple trees and alder windbreaks at different times throughout the growing season

Results and Discussion

Spray tracer distribution - Orchards

The spray distribution for the standard application to small apple trees is shown in Figures 3.

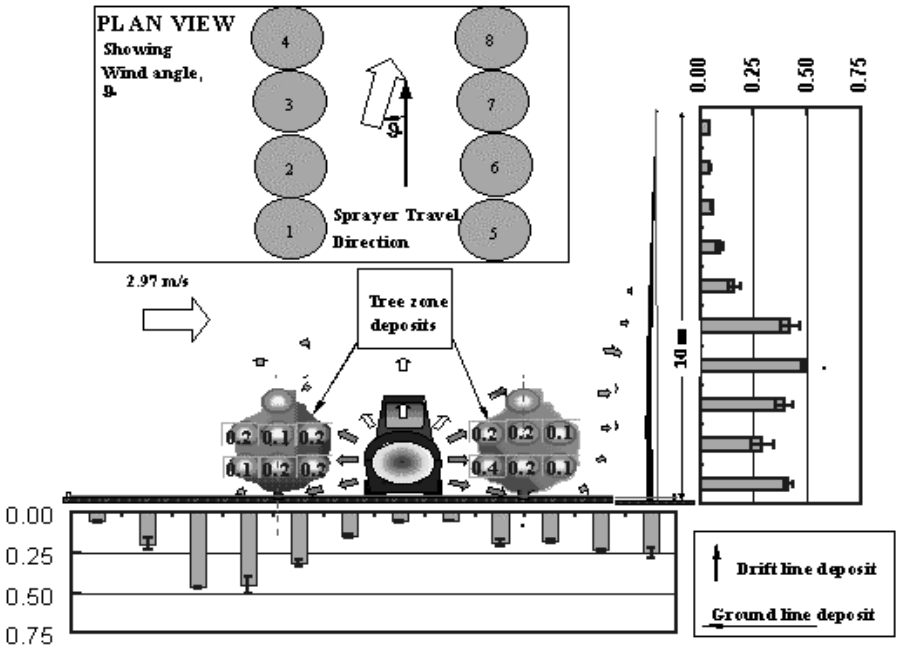

Figure 3 Spray distribution in an orchard of dwarf Cox apple trees on M9 rootstock (2 July 1998). The values of tracer deposit have been normalised by the standard applied volume 300 l/ha

The deposit distribution on the near-ground line collectors shows spray penetration beyond the first row of trees on both sides of the sprayer. The vertical deposit distribution indicates that the spray plume was dispersed to a height greater than 10 m at only 5m downwind of the sprayer centre-line. Above a height of 3.5 m the airborne spray distribution shows the classical exponential decay (HUANG, 1979). Below a height of 3.5 m the spray plume interactions with the crop reduces the local drift line deposit. The profile also shows a local maximum at a height of 0.5 m due to channelling of the spray between the lower branches of the trees and the ground.

The scaling effects of crop structure and applied spray volume

The deposit measurements from the tracer distribution experiments have been used to develop a model to enable the matching of the spray volume output to different trees. Figure 4 presents the leaf deposit and off-target contamination as a series of surface plots against the applied spray volume per unit length, given by the ratio of spray flow rate to sprayer speed (Q/U), and the tree area density (TAD) determined by LIDAR measurements.

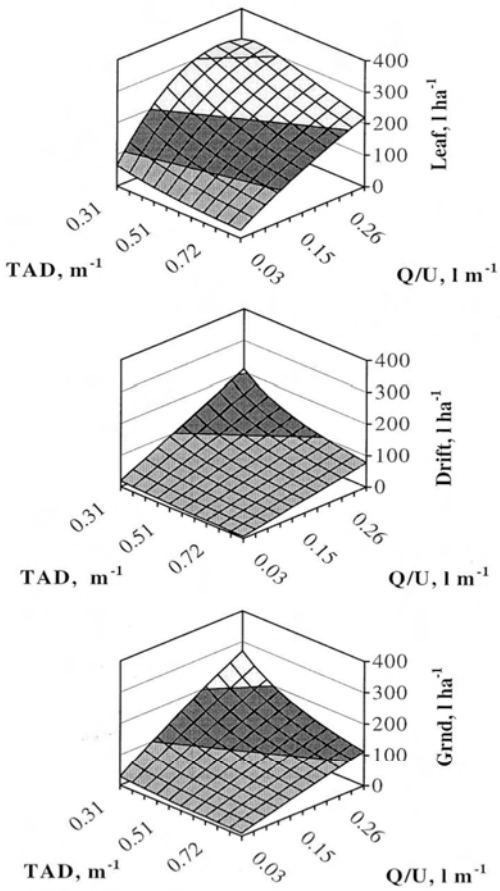

Figure 4 The relationship between the spray volume deposit on different collector surfaces plotted against tree area density (TAD) and the ratio of spray flow rate to sprayer speed (Q/U). The top distribution shows the mean leaf deposit, the middle shows the mean deposit of vertical lines collectors at 5 m downwind of the sprayer centre-line, the bottom shows the mean deposit on horizontal line collectors near the ground

The models show that the all components of tracer deposit increase with the ratio of spray flow rate to sprayer speed (Q/U) and decrease with tree area density (TAD). The surfaces that represent these scaling characteristics are curved to allow for the limiting effect of target saturation, shown in the top distribution of Figure 4 by the flat region where Q/U is high and TAD is low. Furthermore, target saturation conditions increase the off-target contaminants shown in the middle and bottom distributions of Figure 4.

In general, pesticide users should therefore be discouraged from the practice of spraying at or above these target saturation conditions, unless this is a requirement for biological efficacy, because the spraying process becomes inefficient with poor target deposit and increased environmental exposure.

Applied dose reduction to suit crop structure

The leaf deposit model has been reformulated to link TAD with the ratio of applied dose to maximum dose and in this way bring about reductions in drift exposure from orchard with low values of TAD. Figure 5 shows a plot of leaf deposit for constant sprayer operating conditions (i.e. Q/U = 0.08 l m^{-1}). To obtain this plot it has been assumed that the maximum applied dose gives an efficacious deposit on tree crops with a

tree area density of 0.6 m^{-1} and this defines the upper limit shown in Figure 5. The adjustment of applied dose below a TAD of 0.6 m^{-1} has been calculated to give a constant mean leaf deposit and for this condition the applied dose is proportional to TAD. This characteristic is lower limited by the target saturation conditions, in this case for a very low value of TAD below 0.1. However, increasing the spray volume Q/U will raise the lower limit and thus reduce to potential for improving environmental exposure.

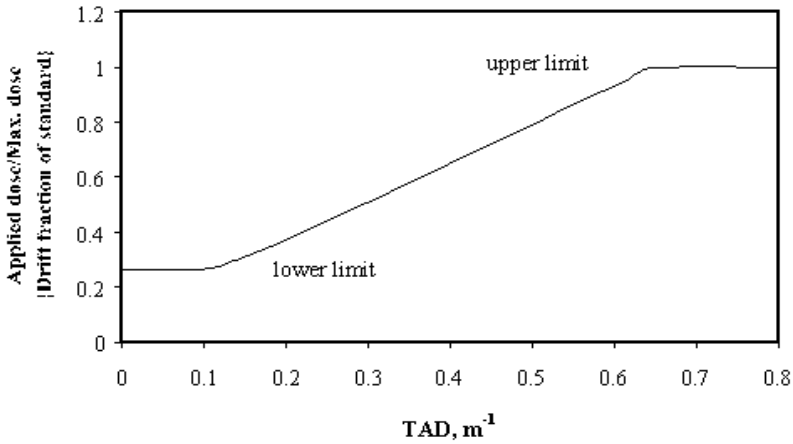

Figure 5 The relationship between applied dose and tree area density for a constant mean leaf deposit. The vertical axis is also interpreted as the fraction of the standard exposure to drift contamination (i.e. at maximum applied dose)

Spray tracer distribution - Windbreaks

Figure 6 shows the contrast between the drift redistribution when the windbreak is open, early in the season, and when the windbreak is very densely foliated, late in the season (i.e. a windbreak area density increases of a factor of three). The results for the low-density windbreak, shown in Figure 6 by the upper pair of distributions, appears to filter the spray drift cloud by allowing it to pass through the windbreak. Hence the drift distributions, before and after the windbreak, have a similar shape with the local maximum at the bottom. He results from the late season measurements show the spray cloud moving up and over the windbreak. Therefore, the downwind drift distribution has a local maximum at a height of 7.5 m (i.e. slightly greater than the maximum height of the windbreak) and the local drift at ground level is lower than at early season. The averages of these distributions suggest that typical reductions of spray drift are in the range 86 – 91% for a 7 m alder windbreak. This is comparable with the improvements that appear to be possible when the standard axial fan sprayer is compared with the best equipment currently available.

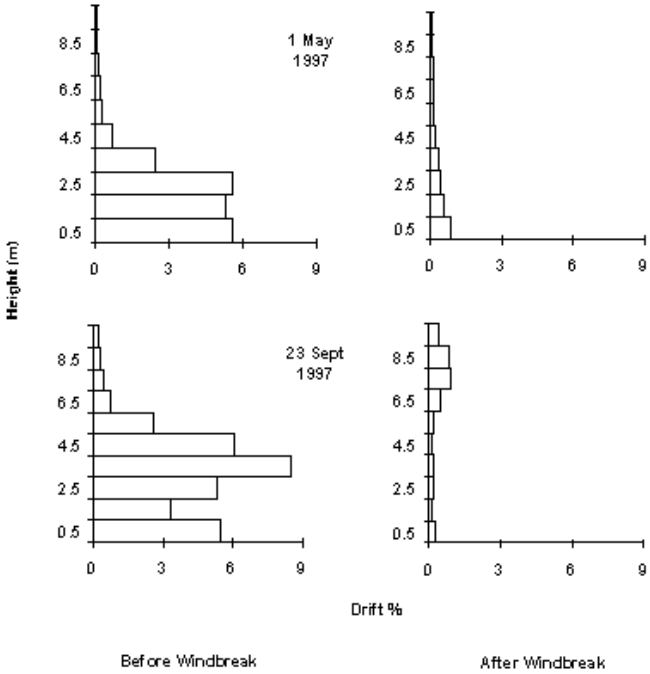

Figure 6 Drift reduction and redistribution due to an alder windbreak at different times during the growing season

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Distribution of drifted substances in Vegetation

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Abstract

Outdoor experiments in field crops, vine cultures and orchards were performed according to good agricultural practice assessing the pesticide exposure of non-target plants. It was demonstrated that not only direct spray drift during the application, but also the atmospheric drift of vaporized pesticides from target plants and soil in the post-application phases caused pesticide contamination in neighbouring ecotones. Particularly semivolatile pesticides like, fenpropimorph, vinclozolin and pyrimethanil occurred in the non-target plants in concentrations exceeding 10% of the concentrations measured in the target plants.

Introduction

According to the German Plant Protection Act (PflSchG 1998) and the Council Directive 91/414/EEC (EC, 1991) of the European Communities (KLEIN & GOEDICKE, 1993) unjustifiable effects on terrestrial ecosystems have to be avoided. Ecotones with their wild species of fauna and flora in the vicinity of agricultural areas have to be protected against pesticide contamination. It is known that arial short-range-transport of pesticides occurs by direct spraydrift during application and by volatilization from the target area in the post-application phase, whereas the degree of contamination for non-target terrestrial ecosystems is unknown (KÖRDEL et al., 1998).

The principal aim of the following studies was to demonstrate whether pesticide applications cause pesticide concentrations in neighbouring non-target ecotones, which could have adverse effects on fauna and flora, and to find out whether hedges in the neighbourhood decrease pesticide drift.

Field Experiments

Field experiments were carried out by herbicide, fungicide and insecticide applications in field crops and in tall growing crops according to good agricultural practice. Depending on the assumed main wind direction the pesticide transport and the exposure scenario were observed on the east, west, north or south wing of the treated field. An overview over the characteristics of the application experiments is given in Table 1a + b. For each field experiment several measuring periods (runs) were performed after each of which soil, air and plant samples were taken (KÖRDEL et. al., 1998). Depending on the size of the treated field the application phase (run 1) lasted up to 2 h; the measuring periods after the application were 2 h and longer.

Table 1a Characteristics of the field experiments – field crops

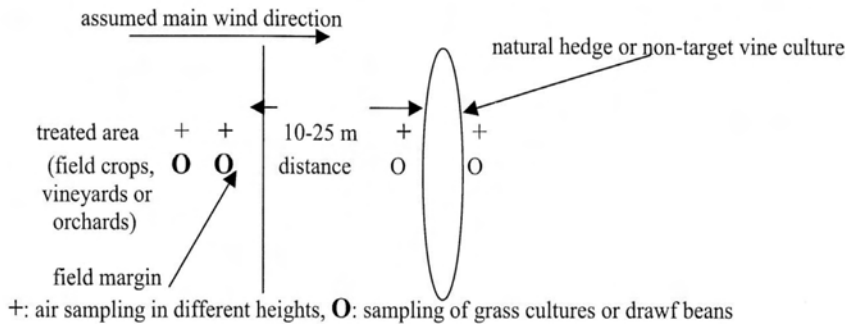
Application	Spring 1995	Summer 1995
Type of nozzles	20 slit-nozzles	20 slit-nozzles
Pressure	1.5 bar	1.5 bar
Droplet sizes	100-500 µm	100-500 µm
Boom height	70 cm above plants	50 cm above ears
Spraying width	10 m	10 m
Speed	8 km/h	8 km/h
Pesticide concentrations	999 g/ha bentazone, 699 g/ha dichlorprop, 800 g/ha chlorothal., 375 g/ha fenpropimorph	100 g/ha parathion, 800 g/ha chlorothalonil
Target plants	barley, 3-5 leaves stage	triticale, in ripeness
Plant cover	80 %	100 %
Target area size	4.5 ha	2 ha
Edge side	130 m	145 m
Wind speed in main wind direction	0.9-1.5 m/s	2.3-3.3 m/s

Table 1b Characteristics of the field experiments - tall growing crops

Application	**August 1996**	**August 1996**	**May 1997**
Nozzles	10 ALBUZ red, yellow	10 ALBUZ yellow	10 ALBUZ yellow
Pressure	13 bar	10 bar	10 bar
Speed	4.0 km/h	5.7 km/h	6.5 km/h
Liquid consumption	204 l/ha	128 l/ha	226 l/ha
Formulations	ME-605	Ronilan	ME-605, Scala, Omnex
Pesticide concentrations	100 g/ha parathion	2124 g/ha vinclozolin	183 g/ha parathion 452 g/ha pyrimethanil 21 g/ha penconazol
Target plants	Vine (2.2 m)	vine (2.2 m)	Apple trees (2.5 m)
Target area size	0.98 ha	0.98 ha	0.44 ha
Edge side	85 m	85 m	110 m
Wind speed in main wind direction	Till 1.22 m/s	Till 0.08 m/s	Till 2.13 m/s

The pesticide concentrations in the air were measured on the treated field, in different downwind distances to the treated field and in different heights (KLÖPPEL et. al., 1997). The climatic conditions, like wind speed, wind direction, air temperature, average relativity humidity etc. were determined. For the single measuring periods the pesticide discharges from the target area in the main wind direction could be estimated by the average wind speed in the main wind direction and the average pesticide concentration at the field edge (KLÖPPEL et. al., 1997).

To assess the pesticide exposure of neighbouring terrestrial ecosystems pesticide concentrations in grass cultures, dwarf beans (biomonitoring) and in the leaves of natural hedges or neighbouring vine cultures situated downwind were measured (Figure 1). The grass cultures or the dwarf beans were placed at different distances to the field, and, if possible, in front of, inside and behind the natural hedges. They were either placed at the sampling sites before application and taken off after run 1, run 2, run 3 etc. or they were placed before each run and taken off immediately after the run. Besides, samples of the target plants were taken after each run for analysis of pesticide concentrations.



Note: +=air sampling in different heights, O=sampling of grass cultures or dwarf beans

Figure 1 Scheme of the drift experiments with sampling points

Results

Pesticides in the air

Figure 2 shows the fenpropimorph concentrations in the air after the different measuring periods (runs 1-4) in different distances to the target area during and after application of barley (3-5 leaves stage) in spring 1995. Besides fenpropimorph the fungicide chlorothalonil and the herbicides bentazone and dichlorprop-p were applied. While the low volatile herbicides bentazone and dichlorprop-p were only found in the air in the spray phase, the more volatile fenpropimorph (vapour pressure: $2.3*10^{-3}$ Pa (20 °C)) occurred in concentrations up to 400 ng/m^3 in the post-application phases. Before and behind the natural hedge at a distance of about 10-20 m to the target area fenpropimorph concentrations of 50-100 ng/m^3 were measured in the post-application phase. The downwind discharges of fenpropimorph and chlorothalonil in the post-application phase were higher than in the spray phase, whereas the calculated total discharges of fenpropimorph and chlorothalonil were lower than 1% of the applied pesticide amount (Figure 3).

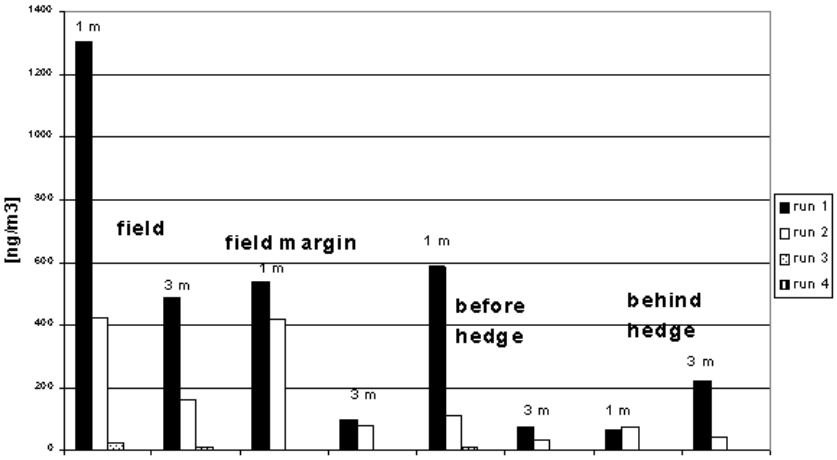


Figure 2 Fenpropimorph concentrations in the air (spring 1995)

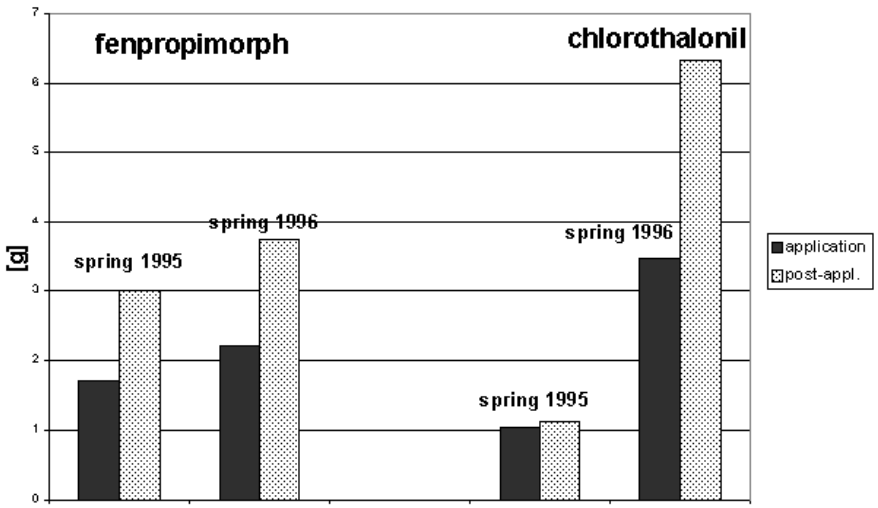


Figure 3 Total downwind discharge amounts from the target area (spring 1995)

Pesticides in non-target plants

Fenpropimorph was taken up in high amounts by downwind non-target plants (grass cultures) during the field experiments in spring 1995. Directly after the application phase fenpropimorph concentrations of 3.5 µg/g were found in the grass cultures at the field margin, conc. of 2.6 µg/g were found in the natural non-leaved hedge and 1.4 µg/g were determined behind the hedge (Figure 4). The fenpropimorph concentrations of the grass cultures placed inside the unleaved hedge were lower at 1 m height than at 3 m height. Behind the hedge fenpropimorph concentrations in the grass cultures increased to 1.4 µg/g 24 h after application (after run 3).

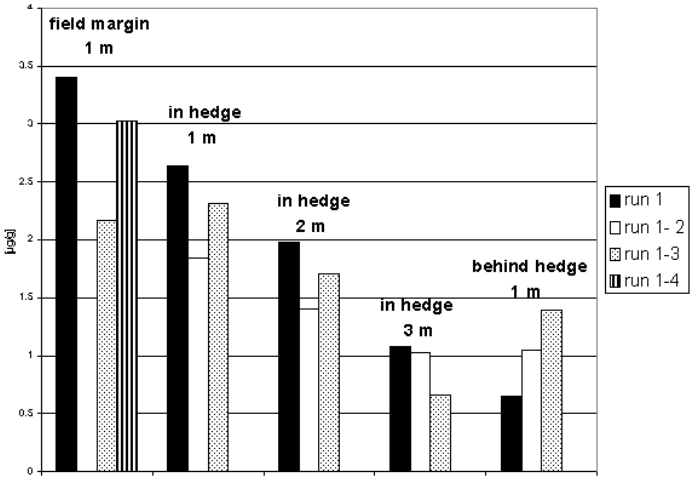


Figure 4 Fenpropimorph concentrations in grass cultures (spring 1995)

The fenpropimorph concentrations in the non-target plants were determined in some cases to more than 10% of the concentrations in the barley (target plants: 12 µg/g) (Figure 5). After run 2, 3 and 4 the fenpropimorph concentrations in the grass cultures decreased slowly, while the fenpropimorph content of the target plants declined rapidly. About 6-10 h after application no differences between the fenpropimorph concentrations in the target plants and in the non-target plants were found (Figure 5).

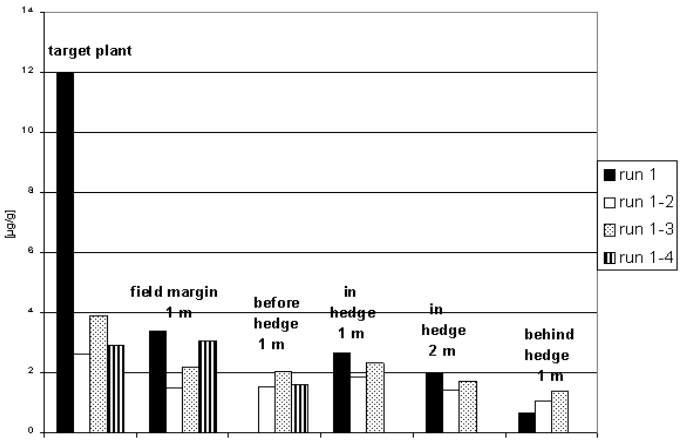


Figure 5 Fenpropimorph concentrations in the target plants and in the grass cultures (spring 1995)

The experiments in spring 1996 show that fenpropimorph was not only taken up by non-target plants in the application phase via direct spray drift, significant amounts were also taken up by volatilization in the post-application phase until day 2 after application by average wind speeds of 0.9-1.5 m/s (Figure 6). At the field margin 0.6 µg/g fenpropimorph, before the hedge 0.3 µg/g fenpropimorph and behind the hedge 0.2 µg/g fenpropimorph were taken up by the grass cultures in run 2, i.e. 0-4 h after the spray phase.

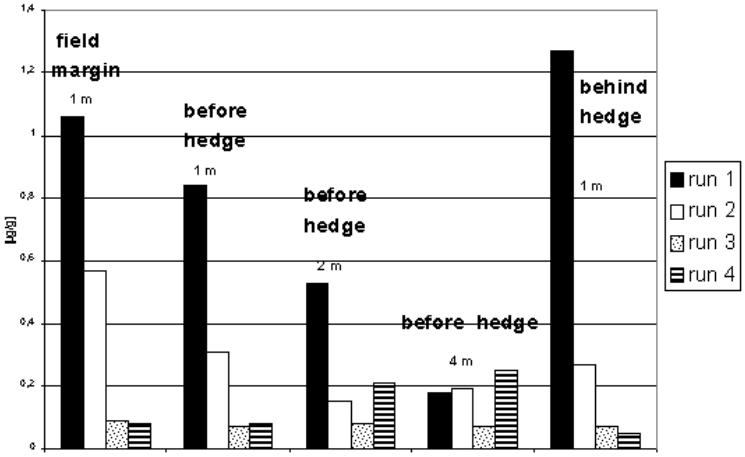


Figure 6 Fenpropimorph uptake from grass cultures in the single runs (spring 1996)

During the application experiments in summer 1995 barley in ripeness was treated with the fungicide chlorothalonil and the insecticide parathion-ethyl. The concentration of the lower volatile chlorothalonil in the grass culture at the field margin amounted to 2.6 µg/g after the application phase and increased to 4.0 µg/g after the second measuring period (run 2), i.e. 5 h after the end of the spray phase (Figure 7). In the grass cultures placed before the leaved natural hedge the chlorothalonil concentrations were higher than in the grass cultures behind the hedge. In the leaves of the natural hedge situated at a 10 m downwind distance to the target area the chlorothalonil concentrations decreased from 1.4 µg/g after run 1 to 0.17 µg/g after run 4. Though the downwind discharge of the more volatile parathion-ethyl was high (about 17% of the applied amount was discharged), the uptake of parathion-ethyl by the grass cultures and by the leaves of the hedge was low. A maximum parathion-ethyl concentration of 0.26 µg/g was found in the grass cultures at the field margin after the application phase (Figure 7). In the leaves of the hedge the parathion-ethyl concentration was below 0.1 µg/g.

The application experiments in vine cultures and orchards showed that pesticide exposure of non-target plants in the spray phase mainly depended on the climatic conditions, like wind speed, wind direction and the inherent properties of the pesticides. Due to the low wind speed (0.62 m/s) the parathion-methyl concentrations in the grass cultures quickly decreased with increasing distance to the target area directly after the application in the vineyard (Figure 8). The grass cultures were placed in 1.5 m, 10 m and 25 m distance to the treated area in the assumed main wind direction, Parathion-methyl (vapour pressure: $2.0*10^{-3}$ Pa) was taken up by the grass cultures not only in the spray phase, but also in the post-application phases. For instance, during one day after application about 50 ng/g parathion-methyl was still taken up by grass cultures at 10 m and 25 m distance. The less volatile fungicide vinclozolin was taken up in higher amounts by the grass cultures at 1.5 m, 10 m and 25 m distance to the target area, though the wind speed in the main wind direction was very low (Figure 8). The maximum measured vinclozolin concentration of 15 µg/g in the grass culture at 1.5 m distance corresponded to about 10% of the concentration in the leaves of the treated vine plants. In the vine leaves of the neighbouring field at a distance of 40 m to the target area, the vinclozolin concentration reached a maximum of 600 ng/g. In the post application phases an uptake of vinclozolin by the grass cultures placed at 10 m and 25 m distance was observed.

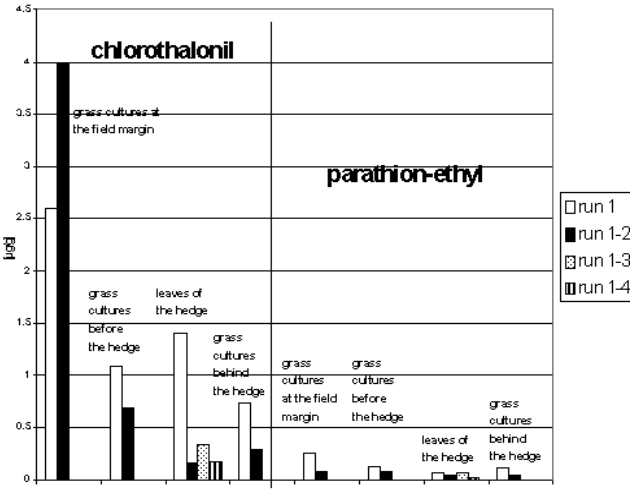


Figure 7 Chlorothalonil and Parathion-ethyl concentrations in grass cultures and in the leaves of the natural hedge (summer 1995)

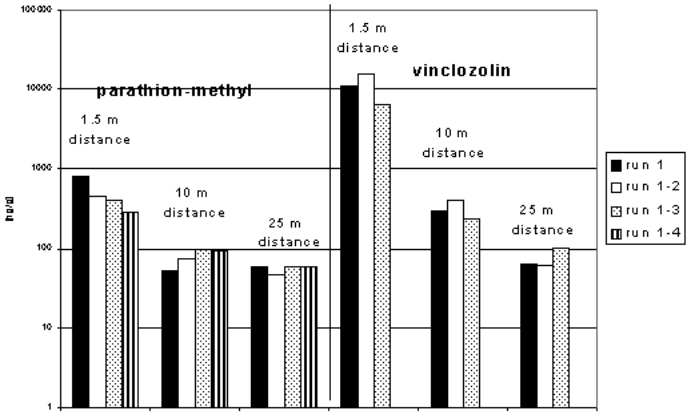


Figure 8 Parathion-methyl and vinclozolin concentrations in grass cultures placed in different distances to the treated vine culture (August 1996)

Figure 9 shows the pyrimethanil and parathion-methyl concentrations in dwarf beans, which were placed at the east wing during the field experiments spraying parathion-methyl, pyrimethanil and penconazol in an apple tree culture. At 10 m and 20 m distance high pyrimethanil concentrations of more than 10 µg/g (10-20% of the concentrations in the target plants) and parathion-methyl concentrations of 5-10 µg/g (10-20% of the concentrations in the target plants) were measured after the application phase due to the high wind speed in the main wind direction (2.13 m/s).The pesticide concentrations in the dwarf beans decreased slowly with increasing time after the application.

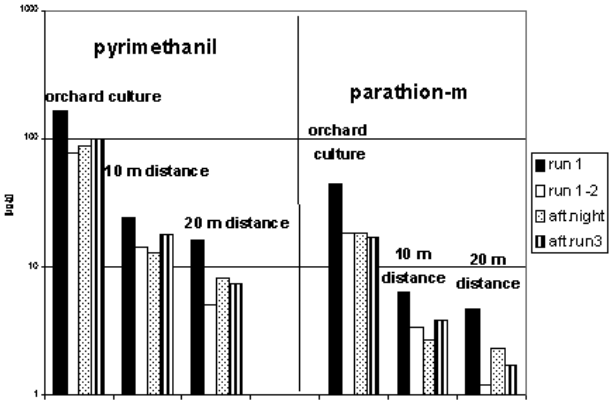


Figure 9 Pyrimethanil and parathion-methyl in dwarf beans (May 1997)

Discussion and Conclusion

The field experiments in crop fields and tall growing crops show that during and after pesticide application according to good agricultural practice significant pesticide concentrations were built up in neighbouring non-target plants. Depending on the climatic conditions, like wind speed, wind direction and air temperature as well as the inherent properties of the pesticides the pesticide concentrations in non-target plants may exceed 10% of the actual pesticide concentration in the target plants. The pesticide uptake in non-target plants occurred not only by direct spray drift in the spray phase but also in significant amounts by atmospheric drift in the post-application phase. A separate measuring of particle-bound and vaporized pesticides was not possible (KÖRDEL et al., 1998). It is assumed that the atmospheric drift mainly consists of pesticides which may evaporate from droplets settled on the plant and soil surfaces.The local pesticide concentration in the air influences pesticide uptake in non-target plants; in field and laboratory experiments simple relationships were found between pesticide concentration in the air and in the plants (KLÖPPEL et al., 1997; BREEZE, 1990; VAN RENDSBURG et al., 1990). On the other hand not only the concentrations in the air but also the inherent properties of the pesticides are responsible for the plant uptake. Highly volatile pesticides are transported from the target area in higher amounts and occurred in the air in high concentrations, however, they are taken up in lower amounts by non-target plants. Generally, the results demonstrate that under realistic application conditions the impact of pesticide drift causes concentrations in neighbouring non-target plants, which cannot be neglected; particularly the semivolatile pesticides with vapour pressures between $5*10^{-3}$ and 10^{-6} Pa have to be considered. The observed shelter effects of downwind hedges were low.

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VAN RENDSBURG, E., BREEZE, V.G. (1990). Uptake and development of phytotoxicity following exposue to vapour of the herbicide 14C 2.4-D butyl by tomato and lettuce plants. Journal of Environmental and Experimental Botany **30**, 405-414.

Current state of the development of drift reducing technique in Germany

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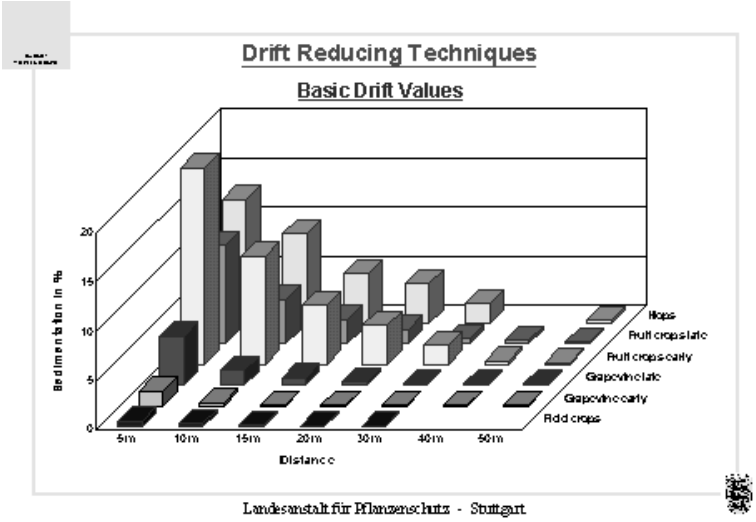
Introduction

Between 1989 and 1992a research program was carried out all over Germany measuring drift during the application of plant protection products in different crops.

The result of this program were these basic drift values, representing the deposition on the soil beside the treated area given in percent of the application rate up to a distance of 50 m.

The height of these values is very different. At the 5m-point e.g. the percentage is far below 1% for field crops, while in the early stage of fruit crops 20% have been measured.

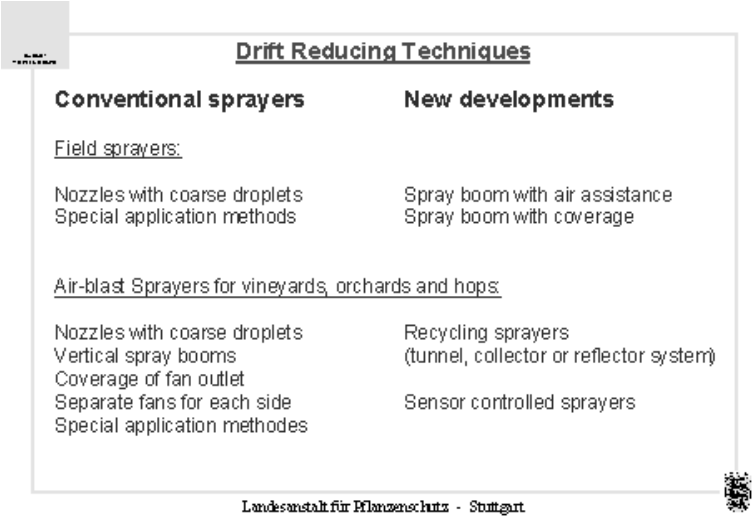
The use of these basic drift values made it necessary to develop techniques and application methods which reduce drift to a lower level.



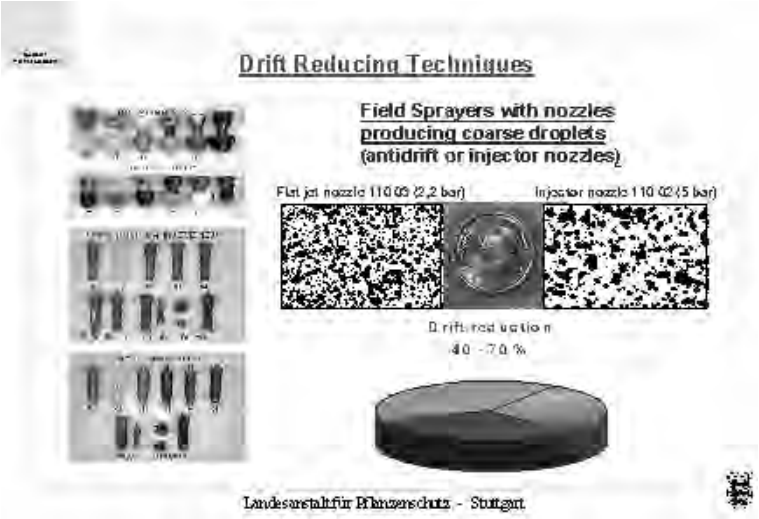
There are two fundamental possibilities for drift reducing techniques. The first one is to modify and improve the conventional sprayers, the second one are new developments. In my opinion the first one is the most important one, because the modification of conventional sprayers including sprayers which are already in use is the only way to get a high number of sprayers which produce low drift very soon.

New developments for drift reduction normally are more expensive and the operation may be more complicated than with conventional sprayers.

The universal recipe for drift reduction with every sprayer is the use of nozzles which produce a coarse droplet spectrum.



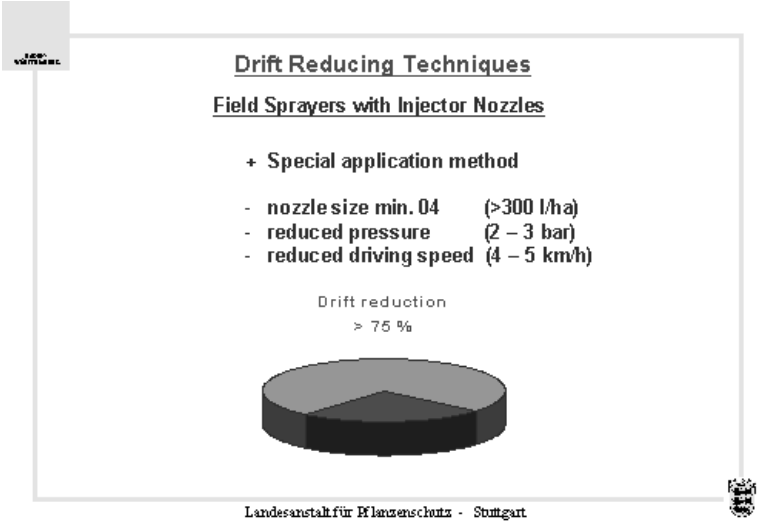
Field Sprayers



The state-of-the-art in nozzle technology are the injector nozzles. Compared with standard flat jet nozzles these nozzles produce very coarse droplets. Additional the percentage of very small droplets is very low.

Compared with the basic drift values, the equipment of field sprayers with injector nozzles instead of the common flat jet nozzles results in a drift reduction between 40 and 70%, depending on nozzle size.

This is relatively low, the reason for it is, that the existing drift values for field crops are already very low.



The drift reduction of field sprayers with injector nozzles can be increased when a special application method is used.

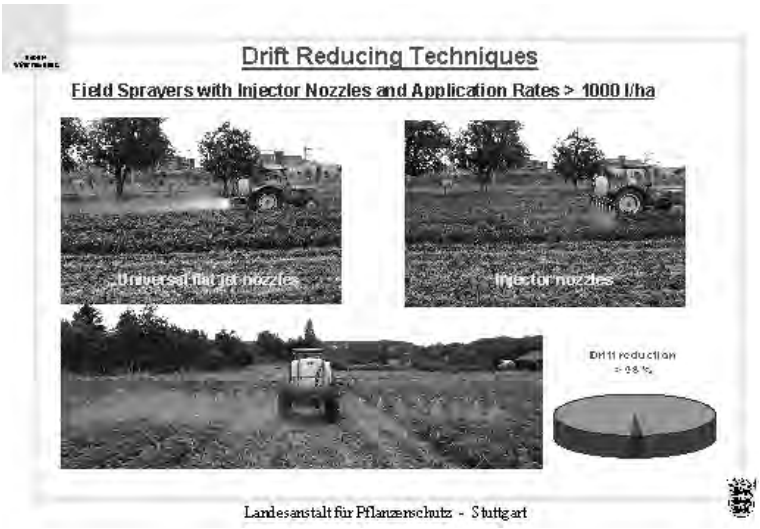
This method excludes the small nozzle sizes below certain size (at the moment 04), the pressure has to be reduced to 2 to 3 bar and the driving speed should not exceed 5 km/h, which means a volume rate of at least 300 l/ha.

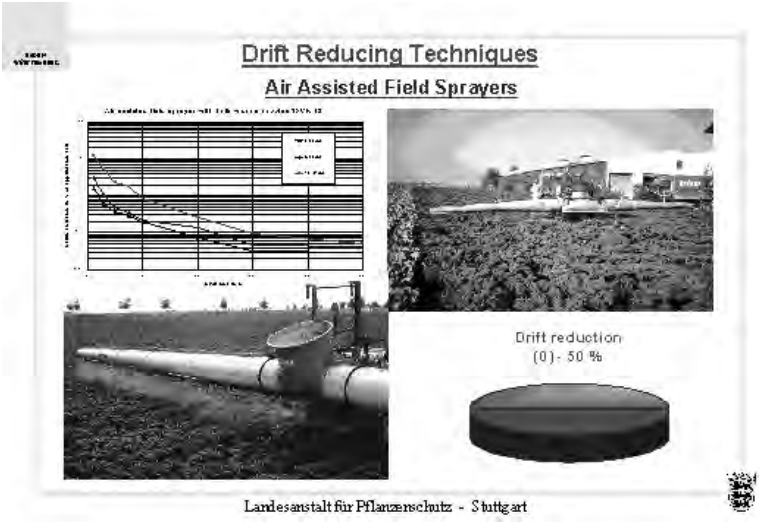
Under these conditions the drift reduction amounts to more than 75%.

Further increasing of nozzle size up to volume rates of 1000 l/ha and more can reduce drift down to less than 2% of the basic drift value for field crops.

Such high application rates sometimes are used in special crops like strawberries.

These photos show a field sprayer spraying strawberries equipped with standard nozzles on the left side and with large injector nozzles on the right side.

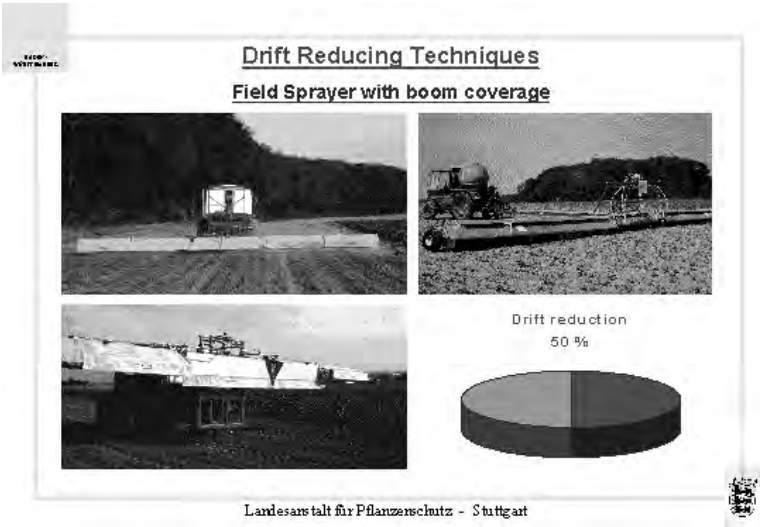




One development of the last years for field crops are air assisted field sprayers. From the beginning it was reported, that drift can be minimized with air assisted sprayers. Therefore sprayers of this type were included in another research program. But with respect to drift reduction the sprayers could not always fulfil the expectations.

The diagram left above shows, that drift possibly increases when air assistance is used with field sprayers and no vegetation or only small plants are present.

The main advantages of air assisted field sprayers are the penetration of dense canopies and good deposition inside the plants. In this case the drift reduction may reach about 50%.

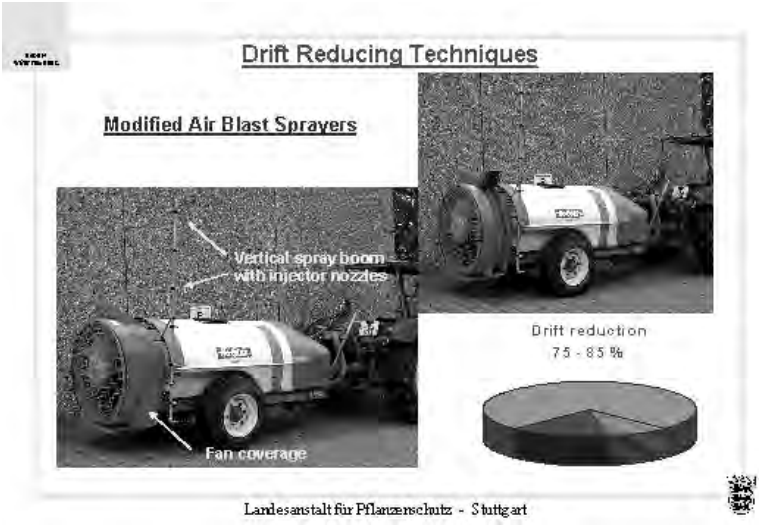


Another development were spray booms with shields to protect the application process against wind influence. Developments from North America as shown right above were not suitable for German conditions.

Therefore one manufacturer produced a boom with a special folding technique for driving on the road.

Of course the boom was more expensive than a standard boom and therefore this type of boom is no longer available on the market in spite of a drift reduction of about 50 % without changing the nozzle type.

Air Blast Sprayers



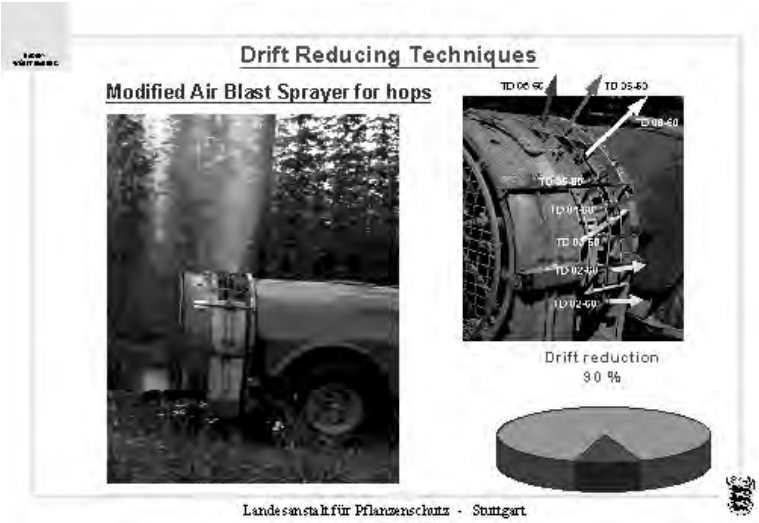
The basic drift values show, that air blast sprayers for vineyards, orchards or hops produce much more drift than field sprayers. They are mainly equipped with hollow cone nozzles which are discharging very fine droplets.

The basis for drift reduction is again the change to coarse droplets.

Additional the fans have to be modified in order that a certain number of rows rows (e.g.5) at the edge of the crop can be sprayed without air assistance in wind or drift direction.

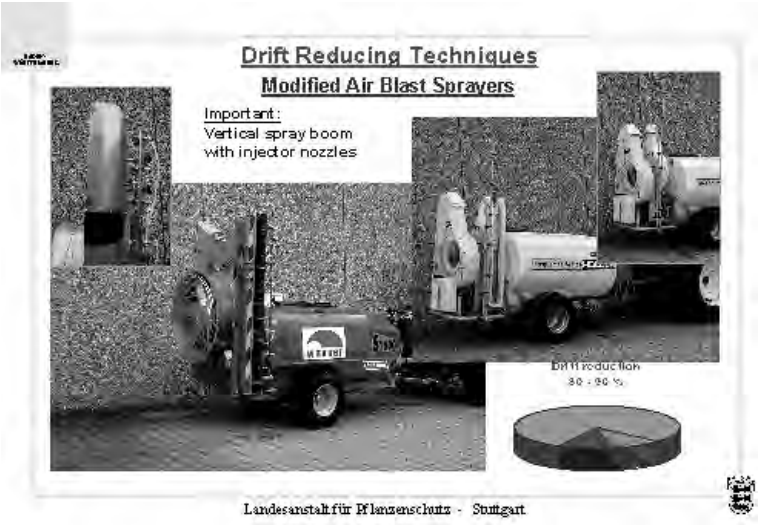
These pictures show a sprayer with an axial fan. The left photo shows it with one side closed by a cover. The complete air volume is directed to the other side. At the closed side a vertical spray boom is mounted for spraying without air assistance.

Latest results of a new research program in fruit crops show a drift reduction of 75 to 85% when several rows are sprayed in this manner.



This spraying method is useful for sprayers with axial fans for hops too.

Spraying the hops from outside with a partly covered fan in combination with injector nozzles with a spray angle of 60 degrees and adapted nozzle sizes as shown in the left photo reduces drift by at least 90%. This spraying method is already accepted very well in practice.

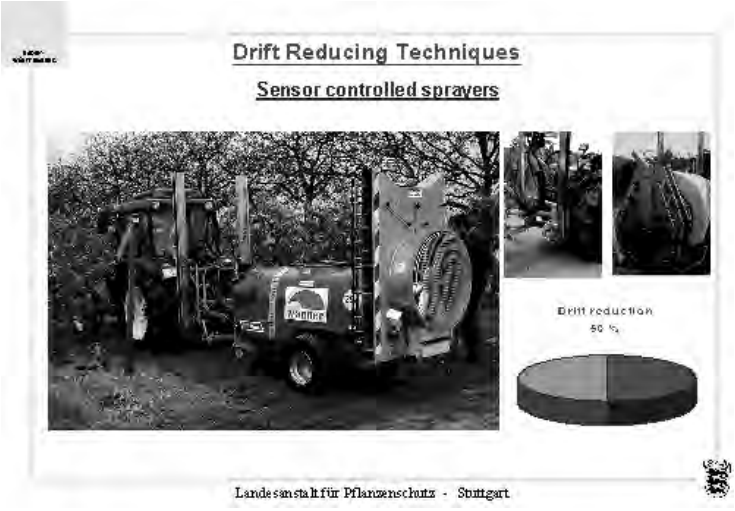


Other conventional air blast sprayers for fruit crops with different fans can be modified according to the same principle.

This includes in every case nozzles with coarse droplets mounted on a vertical boom on the side without air assistance.

The air assistance must be eliminated at one side either by a cover or by changing the air direction. When separate fans for each side of the sprayer are present one of them can be switched off or turned to the other side.

It is expected that sprayers with a nearly horizontal air direction may reduce the drift to a level between 10 and 20% of the actual drift values.

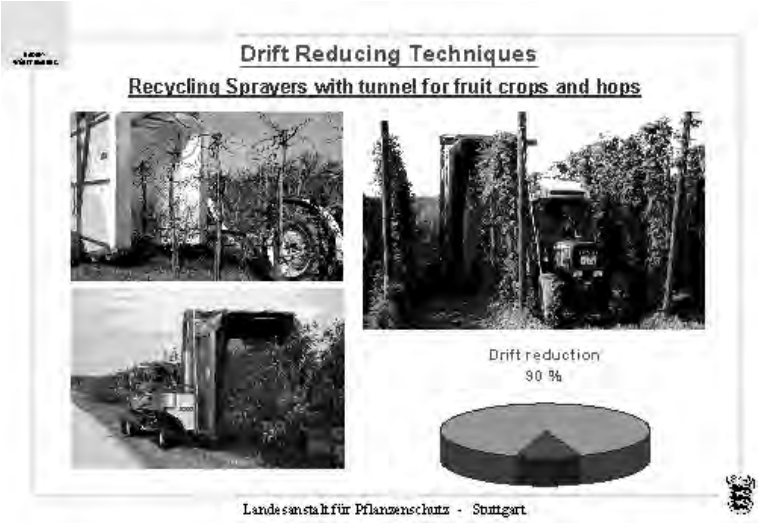


A new development for both vineyards and fruit crops are the sensor controlled sprayers. These sprayers open and close their nozzles according to signals they get from optical sensors. The sensors are able to detect gaps in the sprayed crop. On average this technique may reduce drift up to 50% when common nozzles are used. Increasing this percentage up to 90% should be possible in combination with coarse droplets.

Contrary to conventional sprayers another advantage of sensor controlled sprayers is the saving of plant protection products dependent on the on/off relation of the nozzles.

Another development for vineyards, fruit crops and even hops are the recycling sprayers. They are equipped with tunnels, collectors or reflectors to reduce the influence of the wind onto drift and to save plant protection products by collecting droplets which are not deposited.

The drift reduction exceeds 90% both in vineyards and in fruit crops.



In orchards the use of recycling or tunnel sprayers is reduced to smaller trees depending on height and width of the tunnel.

Tunnel sprayers also can be used in hops, but only when the hop grows up to less than 3,5 m. This was examined during another research project which showed very high drift reduction. But at the moment that is not practice in hop production. During the next decades the hop needs to reach about 8 m for a good yield.

But with regard to drift reduction this is no problem. It is possible to modify conventional sprayer to reduce drift to a very low level. This is valid for hops as well as for nearly every other crop. New developments are necessary but more important are solutions for conventional sprayers which can be realized with low costs in a short time and which are efficient.

Official list of drift reducing technique

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Introduction

Spray technique is an important tool in risk mitigation. In Germany often the authorization of plant protection products cannot be granted or wide buffer zones to surface waters are necessary because of the risks to surface waters. A measure to make the authorization feasible or to reduce the width of the buffer zones is to use a spray technique which reduces the amount of drifted material.

In comparison to the basic drift values which have been determined with common spray techniques the so called "loss reducing equipment" must reduce spray drift fallout significantly. In Germany this has to be proved by field trials. Each successfully tested equipment will then be listed in the official list of drift reducing technique which is published in the federal gazette ("Bundesanzeiger").

If buffer zones are specified in the instruction of the plant protection product a reduced buffer zone width can be specified for applications with loss reducing equipment that is referenced by the official list. Therefor the list offers the opportunity to make use of the benefits of loss reducing equipment in a way that applications are either possible now or that the buffer zone is smaller and a greater part of the field can be treated.

Current state

In 1993 the register "loss reducing equipment" has been introduced in Germany. At that time a minimum drift reduction of 90% had been fixed for equipment to be included in this register. Since then the BBA-approval that proves the suitability of the equipment for the designated purpose is a prerequisite. A declaration according to article 25 of the German Plant Protection Act and an entry in the plant protection equipment list is of course essential, too. Field trials have to be done to prove the drift reduction in comparison to the basic drift values in all measured distances. If all this is fulfilled the equipment is registered as a loss reducing equipment. Today there are tunnel sprayers for orchards and vineyards and a sprayer for hops in the register (Figure 1) but no field sprayers.

Federal Biological Research Centre for Agriculture and Forestry
Register "loss reducing equipment"

Corresponding § 15 article 3 sentence 2 of German plant protection law from 15. September 1986 (BGBl. I S. 1505).
The listed equipment has been registered in the paragraph "Drift"of the register "loss reducing equipment" (Publication.: 17.04.1998)

Number of application	Number of Registration	1. Line: Registration/Typ ab 2. line: model	Field of use	Applicant
G 1250 G 1391	E - 349	TSG - A 010 und 011 und 020 und 021	S, W	JOC
G 1407	E - 434	TSG - N 030 bis 032	S, W	JOC
G 1408	E - 537	TSG - S 040	S, W	JOC
G 1410	E - 471	OSG - N 102 und 103, 112 und 113, 122 und 123, 202,	O	JOC
G 1367	E – 1047	N 42 A 17.01 bis 17.24	H	WAN

The registration has been made on the basis of positive tests concerning drift reducing features. That means, that the equipment must not have been tested completely. In column 2 the number of registration of plant protection equipment list is given. O = Orchards; S = Special crops; W = Vineyards; H = Hops

Figure 1 Actual list of loss reducing equipment

Future developments

As has been shown before there is no entry in the register for field sprayers. Basic drift values in field crops are lower than those in vineyards, orchards and hops. This is one reason that makes it more difficult to obtain a drift reduction of 90%.

In the future a new classification system is planned. Classes of 90%, 75% and 50% drift reduction will allow more differentiated buffer zones. The benefits of new techniques which save plant protection products and reduce the drift will be available to more farmers and will solve a lot of problems concerning the application near surface waters. Field sprayers with air induction nozzles that have been tested in field trials obtained a drift reduction up to 80%. So in some cases an entry in the reduction class 75% may be possible.

Efforts are made to harmonise this classification with the LERAP star rating concept in UK and other European approaches such as in NL.

For the future the procedure for the entry in the register could be as shown in Figure 2. This of course is a first draft and changes are possible.

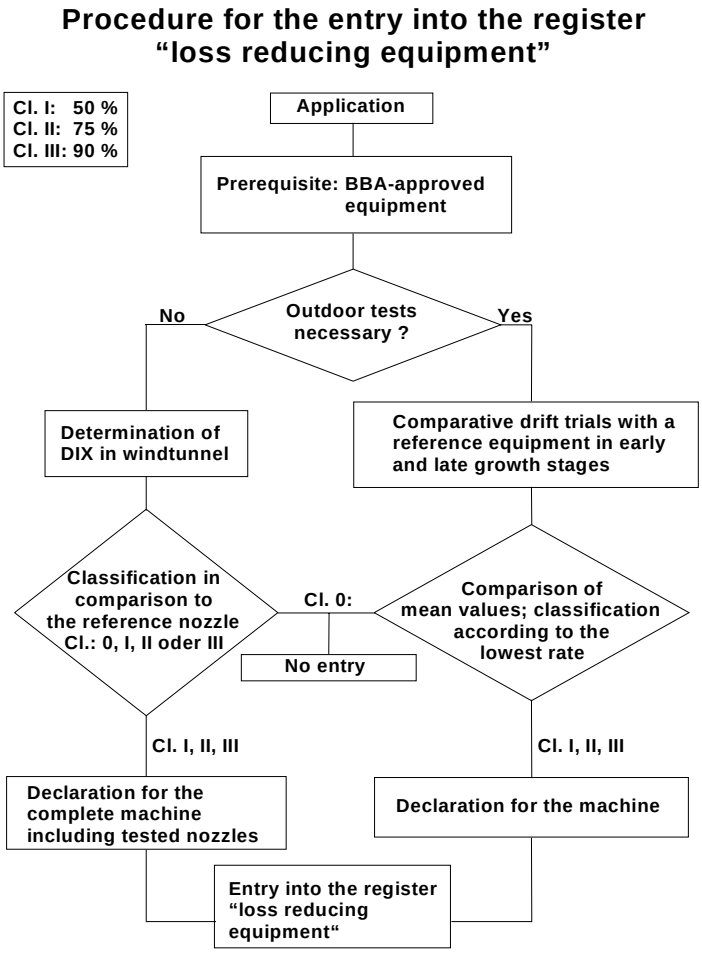

Figure 2 Procedure for the entry into the list of loss reducing equipment

As before an application of the manufacturer and the BBA-approval is a prerequisite. But outdoor tests will not always be necessary. In some cases the determination of the drift potential index (DIX) in a wind tunnel will be sufficient. Future outdoor drift trials will be done in comparison with a predetermined reference equipment. The results of the tests lead to the classification. The tested equipment fits into the class with at least 75% reduction, if all measured values are below the 75%-line. If there is one distance where the measured value is above the 75%-line then the equipment will be classified in the 50%-reduction-class.

Again a declaration according to article 25 and the entry in the plant protection equipment list will be necessary furthermore. After all these requirements are fulfilled the entry into the register will be carried out.

Conclusion

The register "loss reducing equipment" is necessary to make the authorisation of some plant protection feasible or to reduce buffer zone widths. The procedure to test these machines has to be adapted for new developments. Within this process discussions with our european partners are necessary to obtain harmonised regulations.

New basic drift values in the authorization procedure for plant protection products

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Introduction

Since 1995 the assessment of plant protection products with regard to their effects on non-target organisms in Germany has taken place on the base of the drift values published by GANZELMEIER et al., 1995. These values were based on results of 119 drift trials conducted between 1989 and 1992, forming the base for the calculation of the 95th percentile. The basic drift values were obtained by rounding these 95th percentiles up or down to one decimal place.

Improved data base

Besides the above mentioned drift trials additional tests were conducted from 1996 to 1999 for field crops and orchards mainly in accordance with the principles which had been proposed for the earlier tests. Improved analytical methods were used in these experiments, permitting the determination of reliable readings which fall below the previously measurable values by the factor 10. It thus became possible to measure soil sediment values which are smaller than 0.01% of the application rate and which can usually be found at greater distances. That is why the measured range in these experiments was extended up to a distance of 100 m from the treated area, leading to more reliable conclusions on the shape of the drift curve.

All in all, results achieved in 50 agricultural trials and 72 fruit growing trials are now available for evaluation. In viticulture and hops an adoption of new trials for evaluation purposes was not possible yet.

German authorities involved in the authorisation of plant protection products agreed to use the 90th percentile instead of the 95th percentile in future, in conformity with the proposals made by the FOCUS-Surface Water Group (MAUND, 2000).

The extension of trials to measured distances up to 100 m permits the calculation of a equation function by means of a regression analysis which also allows extrapolation at greater distances within certain limits. A power function is suitable as equation function, indicating a linear run of the curve when represented with logarithmic scales on both axes. The general functional equation is:

$$\mathbf{y = a*x^b,}$$

y is in this case the soil sediment expressed in % of a distance x in m. a and b are parameters, to be calculated from the measured values for each crop individually. In orchards and hops one function does not suffice for an adoption, therefore the distances had to be divided into 3 to 10 m and 15 to 250 m. In this case, the height of the plants causes a distinctive lee, leading to a different course of sedimentation as in field crops and viticulture.

In orchards and viticulture a differentiation between early and late growth stages is necessary because of the different drift levels and because there are plant protection products which are only used either in early or in late growth stages.

An extrapolation up to a distance of 250 m was carried through in addition to the distances already measured. Thus, authorisation became possible for some plant protection products so that they may be used in areas which are a long way from surface waters.

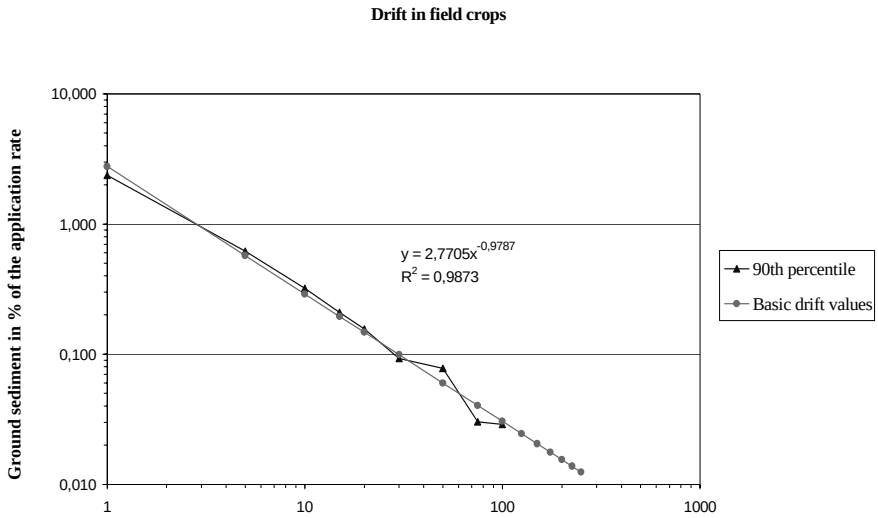

Figure 1: Distance from the treated area in m

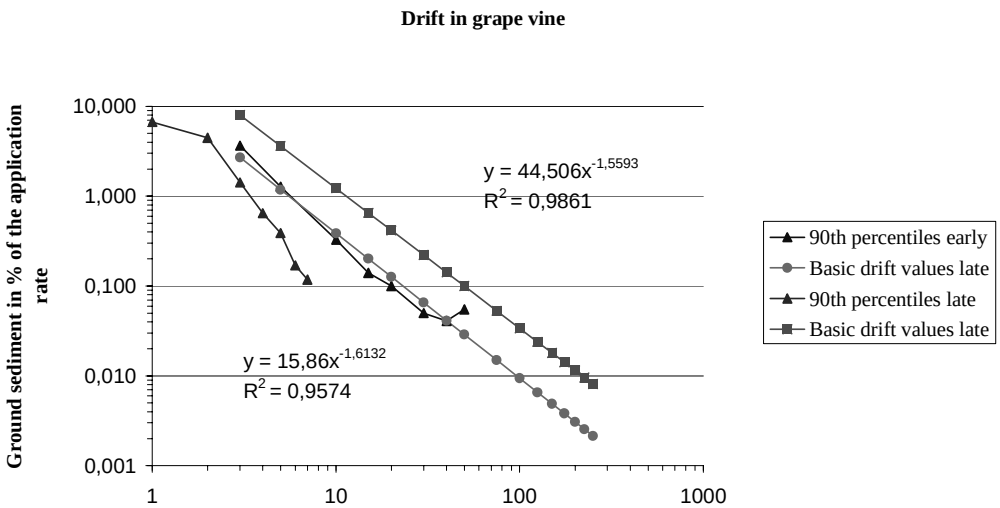

Figure 2: Distance from the treated area in m

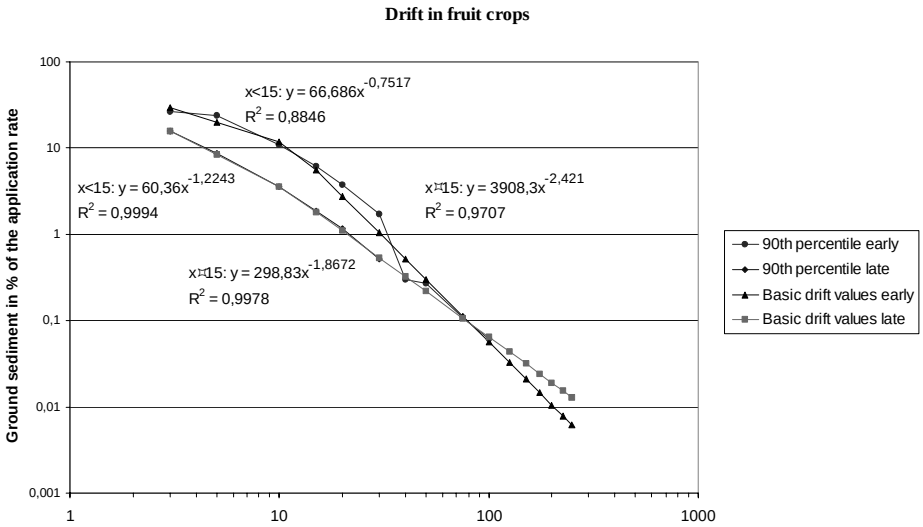

Figure 3: Distance from the treated area

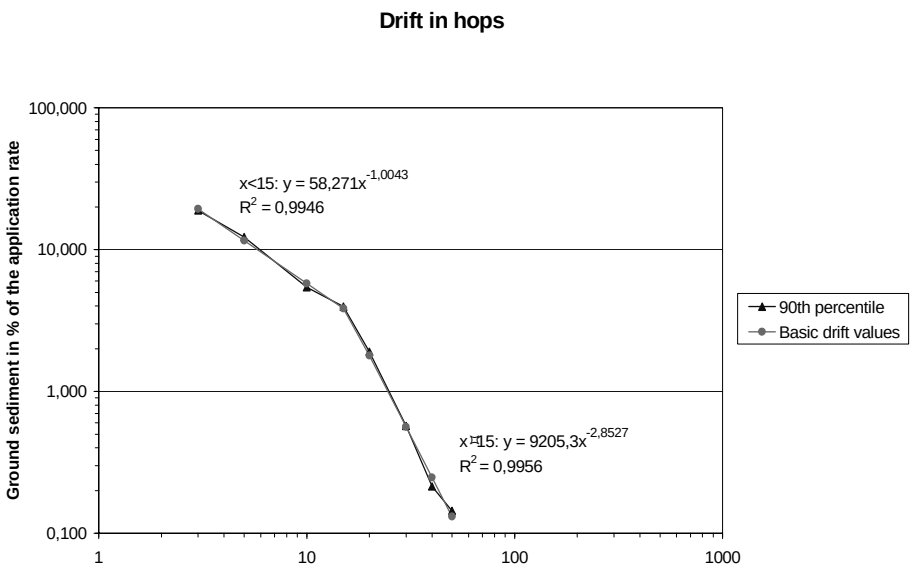

Figure 4: Distance from the treated area in m

Table 1 Parameters of the equation function for different crops

	a	b
Field crops	2,7705	-0,9787
Grape vine early	15,86	-1,6132
Grape vine late	44,506	-1,5593
Fruit crops early, x<15	60,36	-1,2243
Fruit crops early, x≥15	298,83	-1,8672
Fruit crops late, x<15	66,686	-0,7517
Fruit crops late, x≥15	3908,3	-2,421
Hops, x<15	58,271	-1,0043
Hops, x≥15	9205,3	-2,8527

Drift values for multiple uses

At the same time - again in conformity with the FOCUS-group - it was established that a reduced percentile should be used for multiple uses in order not to exceed the 90th percentile cumulatively. The percentiles for multiple applications listed in table 2 represent the exposure for one of the multiple applications, leaving degradation processes aside. If necessary, they can be taken into account later on. Moreover, it was established that risk assessment for multiple uses should at least amount to the Predicted Environmental Concentration (PEC) required for the calculation of a single use. This regulation ensures in the case of fast degrading active substances that a multiple use does not lead to a lower risk than a single use within the assessment.

The following percentiles are used:

Table 2 Percentiles for multiple applications

Number of applications	Percentile used
1	90
2	82
3	77
4	74
5	72
6	70
7	69
8 or more	67

The functional values at a distance of 1, 3, 5, 10, 15, 20, 30, 50, 75, 100, 125, 150, 225 and 250 m were rounded up/down to two decimal places or were this was not sufficient for differentiation, to three decimal places.

The results are thus the following eight tables with basic drift values corresponding to the number of applications.

Conclusions

The new drift values presented in this paper are based on much more results from field trials than the ones published before. Therefore the calculation of a equation function by means of a regression analysis was possible which also allows extrapolation at greater distances within certain limits. These extrapolated values are especially important for regions with a low density of surface waters because a more realistic exposure estimate are now possible. The use of the 90. Percentiles is in accordance with proposals made on EU-level by the FOCUS-group. Additionally this group recommended for multiple uses the use of percentiles depending on the number of applications in order to avoid the the multiplication of worst case situations.

Table 3 Basic drift values for one application - Ground sediment in % of the application rate (90th percentiles)

Distance	**Field crops**	**Fruit crops**		**Grapevine**		**Hops**	**Vegetables Ornamentals Small fruit**	
[m]		**early**	**late**	**early**	**late**		**Height <50 cm**	**Height ≥50 cm**
1	2,77						2,77	
3	0,95	29,20	15,73	2,70	8,02	19,33	0,95	8,02
5	0,57	19,89	8,41	1,18	3,62	11,57	0,57	3,62
10	0,29	11,81	3,60	0,39	1,23	5,77	0,29	1,23
15	0,20	5,55	1,81	0,20	0,65	3,84	0,20	0,65
20	0,15	2,77	1,09	0,13	0,42	1,79	0,15	0,42
30	0,10	1,04	0,54	0,07	0,22	0,56	0,10	0,22
40	0,07	0,52	0,32	0,04	0,14	0,25	0,07	0,14
50	0,06	0,30	0,22	0,03	0,10	0,13	0,06	0,10
75	0,04	0,11	0,11	0,015	0,05	0,04	0,04	0,05
100	0,03	0,06	0,06	0,009	0,03	0,02	0,03	0,03
125	0,025	0,03	0,04	0,007	0,024	0,01	0,025	0,024
150	0,021	0,021	0,03	0,005	0,018	0,006	0,021	0,018
175	0,018	0,015	0,024	0,004	0,014	0,004	0,018	0,014
200	0,016	0,011	0,019	0,003	0,011	0,003	0,016	0,011
225	0,014	0,008	0,016	0,003	0,010	0,002	0,014	0,010
250	0,012	0,006	0,013	0,002	0,008	0,001	0,012	0,008

Table 4 Basic drift values for two applications - Ground sediment in % of the application rate (82nd percentiles)

Distance	**Field crops**	**Fruit crops**		**Grapevine**		**Hops**	**Vegetables Ornamentals Small fruit**	
[m]		**early**	**late**	**early**	**late**		**Height <50 cm**	**Height ≥50 cm**
1	2,38						2,38	
3	0,79	25,53	12,13	2,53	7,23	17,73	0,79	7,23
5	0,47	16,87	6,81	1,09	3,22	9,60	0,47	3,22
10	0,24	9,61	3,11	0,35	1,07	4,18	0,24	1,07
15	0,16	5,61	1,58	0,18	0,56	2,57	0,16	0,56
20	0,12	2,59	0,90	0,11	0,36	1,21	0,12	0,36
30	0,08	0,87	0,40	0,06	0,19	0,38	0,08	0,19
40	0,06	0,40	0,23	0,03	0,12	0,17	0,06	0,12
50	0,05	0,22	0,15	0,02	0,08	0,09	0,05	0,08
75	0,03	0,07	0,07	0,01	0,04	0,03	0,03	0,04
100	0,023	0,03	0,04	0,008	0,03	0,01	0,023	0,03
125	0,019	0,02	0,024	0,005	0,02	0,007	0,019	0,02
150	0,015	0,011	0,017	0,004	0,015	0,004	0,015	0,015
175	0,013	0,008	0,013	0,003	0,012	0,003	0,013	0,012
200	0,012	0,005	0,010	0,002	0,009	0,002	0,012	0,009
225	0,010	0,004	0,008	0,002	0,008	0,001	0,010	0,008
250	0,009	0,003	0,006	0,002	0,007	0,001	0,009	0,007

Table 5 Basic drift values for three applications - Ground sediment in % of the application rate (77th percentiles)

Distance	**Field crops**	**Fruit crops**		**Grapevine**		**Hops**	**Vegetables Ornamentals Small fruit**	
[m]		**early**	**late**	**early**	**late**		**Height <50 cm**	**Height ≥50 cm**
1	2,01						2,01	
3	0,68	23,96	11,01	2,49	6,90	15,93	0,68	6,90
5	0,41	15,79	6,04	1,04	3,07	8,57	0,41	3,07
10	0,20	8,96	2,67	0,32	1,02	3,70	0,20	1,02
15	0,14	5,23	1,39	0,16	0,54	2,26	0,14	0,54
20	0,10	2,36	0,80	0,10	0,34	1,05	0,10	0,34
30	0,07	0,77	0,36	0,05	0,18	0,34	0,07	0,18
40	0,05	0,35	0,21	0,03	0,11	0,15	0,05	0,11
50	0,04	0,19	0,13	0,02	0,08	0,08	0,04	0,08
75	0,03	0,06	0,06	0,01	0,04	0,03	0,03	0,04
100	0,021	0,03	0,03	0,006	0,03	0,01	0,021	0,03
125	0,017	0,015	0,022	0,004	0,02	0,007	0,017	0,02
150	0,014	0,009	0,016	0,003	0,014	0,004	0,014	0,014
175	0,012	0,006	0,012	0,002	0,011	0,003	0,012	0,011
200	0,010	0,004	0,009	0,002	0,009	0,002	0,010	0,009
225	0,009	0,003	0,007	0,002	0,007	0,001	0,009	0,007
250	0,008	0,002	0,006	0,001	0,006	0,001	0,008	0,006

Table 6 Basic drift values for four applications - Ground sediment in % of the application rate (74th percentiles)

Distance	**Field crops**	**Fruit crops**		**Grapevine**		**Hops**	**Vegetables Ornamentals Small fruit**	
[m]		**early**	**late**	**early**	**late**		**Height <50 cm**	**Height ≥50 cm**
1	1,85						1,85	
3	0,62	23,61	10,12	2,44	6,71	15,38	0,62	6,71
5	0,38	15,42	5,60	1,02	2,99	8,26	0,38	2,99
10	0,19	8,66	2,50	0,31	0,99	3,55	0,19	0,99
15	0,13	4,91	1,28	0,16	0,52	2,17	0,13	0,52
20	0,10	2,21	0,75	0,10	0,33	0,93	0,10	0,33
30	0,06	0,72	0,35	0,05	0,17	0,31	0,06	0,17
40	0,05	0,32	0,20	0,03	0,11	0,14	0,05	0,11
50	0,04	0,17	0,13	0,02	0,08	0,08	0,04	0,08
75	0,03	0,06	0,06	0,01	0,04	0,02	0,03	0,04
100	0,019	0,03	0,04	0,006	0,03	0,01	0,019	0,03
125	0,016	0,014	0,023	0,004	0,02	0,006	0,016	0,02
150	0,013	0,008	0,016	0,003	0,014	0,004	0,013	0,014
175	0,011	0,005	0,012	0,002	0,011	0,002	0,011	0,011
200	0,010	0,004	0,010	0,002	0,009	0,002	0,010	0,009
225	0,009	0,003	0,008	0,002	0,007	0,001	0,009	0,007
250	0,008	0,002	0,006	0,001	0,006	0,001	0,008	0,006

Table 7 Basic drift values for five applications - Ground sediment in % of the application rate (72nd percentiles)

Distance	Field crops	Fruit crops		Grapevine		Hops	Vegetables Ornamentals Small fruit	
[m]		early	late	early	late		Height <50 cm	Height ≥50 cm
1	1,75						1,75	
3	0,59	23,12	9,74	2,37	6,59	15,12	0,59	6,59
5	0,36	15,06	5,41	1,00	2,93	7,99	0,36	2,93
10	0,18	8,42	2,43	0,31	0,98	3,36	0,18	0,98
15	0,12	4,61	1,24	0,15	0,51	2,03	0,12	0,51
20	0,09	2,09	0,72	0,09	0,33	0,88	0,09	0,33
30	0,06	0,69	0,34	0,05	0,17	0,29	0,06	0,17
40	0,05	0,31	0,20	0,03	0,11	0,14	0,05	0,11
50	0,04	0,17	0,13	0,02	0,08	0,07	0,04	0,08
75	0,025	0,06	0,06	0,01	0,04	0,02	0,025	0,04
100	0,018	0,03	0,03	0,006	0,03	0,01	0,018	0,03
125	0,015	0,014	0,023	0,004	0,02	0,006	0,015	0,02
150	0,012	0,008	0,016	0,003	0,013	0,004	0,012	0,013
175	0,011	0,005	0,012	0,002	0,010	0,003	0,011	0,010
200	0,009	0,004	0,009	0,002	0,008	0,002	0,009	0,008
225	0,008	0,003	0,008	0,002	0,007	0,001	0,008	0,007
250	0,007	0,002	0,006	0,001	0,006	0,001	0,007	0,006

Table 8 Basic drift values for six applications - Ground sediment in % of the application rate (70th percentiles)

Distance	Field crops	Fruit crops		Grapevine		Hops	Vegetables Ornamentals Small fruit	
[m]		early	late	early	late		Height <50 cm	Height ≥50 cm
1	1,64						1,64	
3	0,56	22,76	9,21	2,29	6,41	14,90	0,56	6,41
5	0,34	14,64	5,18	0,97	2,85	7,79	0,34	2,85
10	0,17	8,04	2,38	0,30	0,95	3,23	0,17	0,95
15	0,11	4,51	1,20	0,15	0,50	1,93	0,11	0,50
20	0,09	2,04	0,68	0,09	0,32	0,83	0,09	0,32
30	0,06	0,66	0,31	0,05	0,17	0,28	0,06	0,17
40	0,04	0,30	0,17	0,03	0,11	0,13	0,04	0,11
50	0,03	0,16	0,11	0,02	0,07	0,07	0,03	0,07
75	0,023	0,05	0,05	0,01	0,04	0,02	0,023	0,04
100	0,018	0,02	0,03	0,006	0,02	0,01	0,018	0,02
125	0,014	0,013	0,018	0,004	0,017	0,006	0,014	0,017
150	0,012	0,008	0,013	0,003	0,013	0,003	0,012	0,013
175	0,010	0,005	0,009	0,002	0,010	0,002	0,010	0,010
200	0,009	0,004	0,007	0,002	0,008	0,002	0,009	0,008
225	0,008	0,003	0,006	0,002	0,007	0,001	0,008	0,007
250	0,007	0,002	0,005	0,001	0,006	0,001	0,007	0,006

Table 9 Basic drift values for seven applications - Ground sediment in % of the application rate (69th percentiles)

Distance	Field crops	Fruit crops		Grapevine		Hops	Vegetables Ornamentals Small fruit	
[m]		early	late	early	late		Height <50 cm	Height ≥50 cm
1	1,61						1,61	
3	0,55	22,69	9,10	2,24	6,33	14,63	0,55	6,33
5	0,33	14,45	5,11	0,94	2,81	7,60	0,33	2,81
10	0,17	7,83	2,33	0,29	0,94	3,13	0,17	0,94
15	0,11	4,40	1,20	0,15	0,49	1,86	0,11	0,49
20	0,08	1,99	0,67	0,09	0,31	0,81	0,08	0,31
30	0,06	0,65	0,30	0,05	0,16	0,26	0,06	0,16
40	0,04	0,29	0,17	0,03	0,10	0,12	0,04	0,10
50	0,03	0,16	0,11	0,02	0,07	0,06	0,03	0,07
75	0,023	0,05	0,05	0,01	0,04	0,02	0,023	0,04
100	0,017	0,02	0,03	0,006	0,02	0,01	0,017	0,02
125	0,014	0,013	0,017	0,004	0,017	0,005	0,014	0,017
150	0,012	0,008	0,012	0,003	0,013	0,003	0,012	0,013
175	0,010	0,005	0,009	0,002	0,010	0,002	0,010	0,010
200	0,009	0,003	0,007	0,002	0,008	0,001	0,009	0,008
225	0,008	0,003	0,005	0,002	0,007	0,001	0,008	0,007
250	0,007	0,002	0,004	0,001	0,006	0,001	0,007	0,006

Table 10 Basic drift values for more than seven applications - Ground sediment in % of the application rate (67th percentiles)

Distance	Field crops	Fruit crops		Grapevine		Hops	Vegetables Ornamentals Small fruit	
[m]		early	late	early	late		Height <50 cm	Height ≥50 cm
1	1,52						1,52	
3	0,52	22,24	8,66	2,16	6,26	13,53	0,52	6,26
5	0,31	14,09	4,92	0,91	2,78	7,15	0,31	2,78
10	0,16	7,58	2,29	0,28	0,93	3,01	0,16	0,93
15	0,11	4,21	1,14	0,14	0,49	1,82	0,11	0,49
20	0,08	1,91	0,65	0,09	0,31	0,78	0,08	0,31
30	0,05	0,62	0,29	0,04	0,16	0,25	0,05	0,16
40	0,04	0,28	0,16	0,03	0,10	0,12	0,04	0,10
50	0,03	0,15	0,11	0,02	0,07	0,06	0,03	0,07
75	0,022	0,05	0,05	0,009	0,04	0,02	0,022	0,04
100	0,017	0,02	0,03	0,006	0,02	0,01	0,017	0,02
125	0,013	0,012	0,017	0,004	0,017	0,005	0,013	0,017
150	0,011	0,007	0,012	0,003	0,013	0,003	0,011	0,013
175	0,010	0,005	0,009	0,002	0,010	0,002	0,010	0,010
200	0,008	0,003	0,007	0,002	0,008	0,001	0,008	0,008
225	0,007	0,002	0,005	0,001	0,007	0,001	0,007	0,007
250	0,007	0,002	0,004	0,001	0,006	0,001	0,007	0,006

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Résumé

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New developments of the methods currently used for risk assessment

Experts of seven countries joined the BBA-workshop. New concepts improving currently used procedures for risk assessment were presented. The approach of probabilistic risk assessment, developed mainly in the framework of ECOFRAME, a project initiated by EPA, was broadly accepted as a good approach of higher tier risk assessment to consider the probability of an effect in an appropriate way.

The current situation in different EU Member States

A comparison of the risk assessment for aquatic systems turned out, that in the United Kingdom, the Netherlands, Austria and Germany generally the same toxicity data are used. For exposure assessment in the United Kingdom, Austria and Germany the german spray drift data are used, whereas in the Netherlands different data are used.

Concerning measures for risk mitigation, differences between the different Member States were obvious, where in the Netherlands the highest degree of differentiation was reported. In the United Kingdom the LERAP concept is already established, allowing for a differentiation with respect to field rate, width of waters and application technique.

Discussions on terrestrial ecotoxicology, especially on non-target athropods, stressed that in the Member States of the EU the principles as laid down in the SETAC/ESCORT Guidance Document are implemented, which according to the BBA serves as a basis for the german authorization procedure as well. No spray zones based on spray drift values, as applied for risk mitigation for aquatic systems, are not implemented. Risk mitigation measures are currently limited to the labelling of pesticides. However, in the United Kingdom fixed buffer zones of 6 m are established in cereals for the use of organophosphates and pyrethroids.

Monitoring

Pesticides can be found in waters by anaytical means and effects on aquatic organisms could be found in a number of studies. However, the magnitudes of the concentrations measured and the effects found matched the scales predicted in the risk assessment only for a short period of time.

Studies conducted in the United Kingdom and in Germany demonstrated significant reductions in the abundance of arthropods after the application of insecticides within the treated fields, however, these studies demonstrated a rapid recolonization as well. In contrast, studies on the effects of spray drift of insecticides into field margins conducted in Germany did not show any significant adverse effects on non-target arthropods.

Differentiation of buffer zones

In the United Kingdom and in Germany proposals for a differentiation of buffer zone restrictions are currently available, introducing the type and vegetation of waters. In addition data to establish simple GIS-systems are available. In the United Kingdom and the Netherlands further data are published, which demonstrate a significant reduction of spray drift by the filtering capacity of the crop and by wind breaks.

The results presented at the workshop support the BBA concept of risk mitigation for terrestrial non-target organisms, which is based on a local risk assessment taking into account landscape characteristics.

Application technique

A number of different new drift reducing techniques are currently being tested. In Germany new techniques are listed in an official list for drift reducing techniques. A general recommendation was to bring together the numerous data on spray drift collected in the United States of America, the United Kingdom, the Netherlands and Germany, in order to construct a uniform data base.

Outlook

The workshop indicated there is a broad consensus of opinion in favour of implementing new tools into risk assessment for both aquatic life and terrestrial life, such as the probabilistic approach, once fully available. Furthermore the aspects of recolonization and recovery together with the relevant landscape features are to be appropriately addressed in risk assessment and risk mitigation. To achieve these goals further work and a follow-up workshop was strongly supported by the participants of WORMM.

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