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LANDSCAPE AND MITIGATION FACTORS IN AQUATIC ECOLOGICAL RISK ASSESSMENT.

Volume 2. Detailed Technical Reviews

Final Report of the FOCUS Working Group on Landscape and Mitigation Factors in Ecological Risk Assessment

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INTRODUCTION TO VOLUME 2

The main findings and recommendations of the FOCUS Work Group on Landscape and Mitigation Factors in Aquatic Ecological Risk Assessment are presented as Volume 1 to this report. The purpose of this second volume is to provide the detailed review material underpinning discussions within the Work Group.

The Work Group reviewed a large amount of technical information within four main topic areas for which subgroups were established. The subgroups were categorized as follows:

- Risk mitigation
- Exposure modelling
- Landscape analysis
- Ecology

Each of these subgroups has produced a detailed technical review for its subject area. The scope of the review exercise that was conducted is provided below along with the relevant section in this second volume of the report:

Risk mitigation	**Section 1**
Current practice in risk mitigation within the framework of 91/414/EEC	1.1
Options to mitigate exposure via spray drift	1.2
Options to mitigate exposure via surface runoff	1.3
Options to mitigate exposure via drainflow	1.4
Mitigation measures applying to all routes of exposure	1.5
Exposure modelling	**Section 2.1 & 2.2**
Refinements to FOCUS Step 3 surface water modelling (Step 4 calculations): edge of field modifications; incorporating mitigation measures; more complex modelling	2.1.1
Modelling at the catchment scale	2.1.2
Probabilistic risk assessment	2.1.3
Use of monitoring data in exposure assessment	2,2
Landscape analysis	**Section 2.3 & 2.4**
Unit of analysis	2.3.1
Site selection process	2.3.2
Landscape factors for higher tier exposure assessment	2.3.3
Exposure estimates for higher tier assessment	2.3.4
Relating landscape factors to a larger area	2.3.5
Supporting information for higher tier exposure assessment	2.3.6
Evaluating the spatial distribution of results	2.3.7
Use of remotely-sensed data in landscape characterisation	2.3.8
Data layers and contacts for spatial analyses	2.4
Ecology	**Section 3**
Overview of current legislative background and protection aims	3.1
Factors that influence organism composition	3.2
Abiotic and biotic factors that influence effects	3.3
Ecological factors that influence exposure	3.4
Landscape factors that influence effects and recovery	3.5

1 REVIEWS OF THE STATE-OF-THE-ART IN MITIGATING RISK

1.1 Introduction

1.1.1 Scope of the review

A broad view of risk mitigation measures is taken and these are defined as all measures and conditions which mitigate risk compared with the standard use situation considered during risk assessment in accordance with the Uniform Principles. This means that not only active mitigation such as implementation of a no-spray buffer zone, but also the absence of a vulnerable situation (e.g. large and/or flowing water bodies with large dilution potential) are considered within the review.

Risk mitigation measures are related to “risk” and therefore label phrases connected to substance-inherent properties are not discussed. Mitigation of point sources of contamination is not considered because no regulatory risk assessment is conducted for this type of exposure. In general, this type of exposure should be reduced to zero by technical means independent of the risk to aquatic organisms. The approach should be comparable with industrial chemicals.

The chapter is divided into major sections which describe current procedures for risk mitigation across Europe, set out the scientific background for risk mitigation measures currently used and explore those measures on the “single field/small water body” scale which are currently not used for regulatory purposes but might be used in the near future.

The main purpose of this report is to identify potential risk mitigation measures from a scientific point of view but considerations related to the implementation in practical agriculture will also be tackled. However, issues related to the acceptance of risk mitigation measures by farmers and especially legal aspects connected with enforceability of restrictions will not be discussed extensively because they are very much related to the specific situation in individual Member States. Ideally, the following overview should give ideas on how to improve risk mitigation measures currently used in order to come to better harmonized approaches which would ease decision-making at the European level.

1.1.2 General comments on implementing risk mitigation measures

The setting of restrictions is not only based on scientific considerations. Legal and administrative requirements, enforceability and acceptance by stakeholders are additional issues to be considered and in practice often as important as scientific problems. The description of the additional scenarios/wording of the restriction is in some member states very much driven by legal requirements to make sure that punishments are possible if users do not follow the restrictions. As the state of the art develops, new phrases must be introduced and if restrictions are on the label it is usually difficult to change all authorizations containing the restrictions at the same time due to legal and administrative requirements. Therefore safety phrases should be as short as possible and should only contain product-specific information. Different phrases for the same problem are usually confusing for users and reduce acceptance amongst them. It is reasonable to refer in safety phrases on the label to official publications where different scenarios are decribed in order to ease the implementation of progress of the state of the art, avoid changes of all authorizations and confusion amongst farmers. All information which is needed to follow risk mitigation measures but which is not product-specific should be included in such documents. These publications can be amended when needed and authorizations are always up to date without any administrative work on single authorizations or Annex I inclusions.

It must be possible under practical conditions to enforce restrictions. Therefore risk mitigation measures which are too sophisticated may be possible from a scientific point of view but not in practice. On the other hand, overly simplistic restrictions (which may be too protective in some situations) are not accepted by farmer organizations because they lead to unjustified limitations on food production.

It is very important to develop risk mitigation measures in close contact with stakeholders (farmers, industry, NGOs etc.) to increase their acceptability. Training courses for farmers by the extension service are especially needed if farmers should follow more difficult to understand risk mitigation measures as for example those currently used in Sweden or UK. Simple computer-based decion-making programmes should be developed. Environmental issues may also be included in the stewardship programmes of companies. In Germany, state authorities developed such programmes and made them available to farmers via the internet. Furthermore, slides and background information were developed which have been used by state authorities to educate farmers. Also in Germany, GIS-based maps have been generated to show which areas have a high recovery potential for terrestrial life. Safety phrases are connected to these maps/lists and farmers can get access to this information via the internet.

Book-keeping is a reasonable tool to ease control of restrictions especially more complicated ones.

This report is mainly focused on protection of the aquatic compartment. Contradictions with other measures to protect terrestrial life or to reduce intake via different exposure routes should be avoided. For example, shrubs and bushes reduce the intake via spray drift but to a much lesser extent via runoff compared with a grassed buffer. In other areas like for example the protection of terrestrial life or drinking water risk mitigation measures are also set. All these approaches must be complementary. Furthermore, it should be considered that a risk mitigation measure in one area like for example a hedgerow to reduce spray drift might cause problems in another like for example the terrestrial area where arthropods are to be protected. Unsprayed crops in buffer zones might cause more problems with pests and result in the need for higher application rates in the rest of the field.

Farmers must not only follow regulations when using plant protection products. Other requirements are set in the frame of nature conservation, fertilizers, liquid manure, soil conservation etc. Attempts should be made to come to integrated measures. In any case, contradictions between these regulations must be avoided. Therefore it is advisable to discuss risk mitigation measures with experts from these other areas before practical use. Setting of environmental quality standards by the food industry has increased especially after the BSE-crisis. Harmonization could be fostered by certification systems like the one in The Netherlands where farmers accept in a contract to follow special requirements regarding the environment.

Another issue to be considered is the relationship between EU, national, regional and local level. If in general the final aim of developing risk mitgation measures is to link restrictions to the risk prevailing under local conditions it is important that risk phrases on the label offer flexibility on the local scale. Clearly, the options for local risk management must be defined on a higher level to ensure consistency of decision-making and the appropriate level of protection.

1.2 Risk mitigation measures currently used in EU-member states to protect aquatic life in the authorization procedure of plant protection products

A review of current procedures for setting risk mitigation measures in the EU Member States was undertaken by direct survey supplemented by review of published papers and presentations to conferences and workshops. The results of this review are summarised in Table 1.1. It should be clearly noted that the summary of risk mitigation measures may not be complete.

Although it was intended to collect data for both aquatic and terrestrial compartments, it turned out that most of the available information is for the former. With respect to the exposure routes, most efforts to date have been made to mitigate entry into surface water via spray drift whereas only preliminary information is available for runoff and drainage. Mitigation of risk arising from surface water exposure via spray drift thus makes up the largest part of the review.

Table 1.1. Summary of risk mitigation measures currently used in Member States.

	Austria	Denmark	Finland	France	Germany	Greece	Ireland	Italy	Netherlands	Portugal	Spain	Sweden	UK
Definition of water body	Not mentioned	Not mentioned	Water reservoir having water the whole year	Not mentioned	All water bodies except those falling dry over long periods of the year	Not mentioned	All water bodies	Not mentioned	All water bodies	All water bodies except those falling dry over long periods of the year	Not mentioned	All water bodies	All water bodies
Main Exposure route	Spray-drift (German drift values)	Spray drift (German drift values)		Runoff, Spray drift	Spray-drift but also drainage and runoff; recently also volatilization	Runoff	Spray-drift (German spray-drift values)	Spray-drift (German drift values) and runoff	Spray-drift; dutch drift values are used for setting mitigation measures	Spray-drift (German spray-drift values)		Spray drift	Spray drift German spray-drift values) but also drainage
Risk Mitigation Measures	Buffer zones up to 50 m	Buffer zones up to 20 m in field crops, 50 m in tall growing crops 30 m in vegetables	Buffer zones on the base of inherent toxicitity; 10 to 25 m distance	Risk mitigation measures are set in the authorization procedure; on catchment level mitigation measures are set to reduce runoff if water quality criteria not met; Diagnosis system to identify problem areas; Implementation of buffer zones and hedgerows	Buffer zones to reduce spray-drift currently up to 20 m; grassed buffer zones and no-tillage techniques to reduce runoff; phys-chem. properties and time of application are used to reduce intake via drainage;	Need of a buffer zone included in the label as a general requirement; further recommendations on regional scale	Buffer zones up to 50 m; 1 and 5 m in arable crops are common	Buffer zones up to 20 m	0.2 -14 m buffer zones (maintained as no-crop zones); lower reduction of spray deposit with distance than German Spray-drift values; recommend-ation of grassed buffer zone and minimum tillage to reduce runoff;	No spray buffer zone, by crop (up to 5-40 m). Recommend-ation on use of low spray drift nozzles; Recommend-ation of grassed buffer zones and minimum tillage.	Unsprayed buffer zones on the label (up to 5-50 m);grassed buffer zones and application timing to reduce runoff. Regarding rice retention time, maximum dilution factor for receiving water body; time to release can be fixed	Buffer zones of 1 m to ditches and 6 m to lakes and streams; alternative is a 10 m strip set aside land adjacent to the water body to get EU-financial support; setting not risk baesed; additional buffer zones to reduce spray-drift;	Buffer zones of up to 5 and 50 m in arable and tall growing crops respectively; regarding drainage period for application is set

	Austria	Denmark	Finland	France	Germany	Greece	Ireland	Italy	Netherlands	Portugal	Spain	Sweden	UK
Local conditions considered	Application rate; Spray-drift reducing technique; Type of water body; Vegetation between application area and water body; Special type of application (band spraying etc.)	Not mentioned	Not mentioned	Selection of active substances with adequate properties; Weed control by rotation; conservation tillage); Treatment period change	Spray-drift reducing technique; Regarding runoff conservation tillage or detention ponds; slope Drainage: Soil type	Not mentioned	Dry ditches;	Application rate; technique; Spray-drift reducing	Spray-drift reducing application technique Windbreak of trees	Not mentioned	Spray-drift reducing technique	Spray-drift reducing technique; application rate; windspeed; wind direction; field size; temperature, spray-quality, boom height; small ditches	LERAP; Spray-drift redcing technique, application rate, type of water body; windbreaks for orchards additionally; (not relevant for pyrethroids, organophosphates) Dry ditches
Enforc-ement	No clear information	No clear information	Within subsidie programmes	Regional pesticide groups supervise actions	State authorities responsible; intensity depends on priority setting in states; farmers may be punished by fees up to 50000 €	Not mentioned	Not mentioned	Not mentioned	No-crop zones to ease enforcement; Inspection service of Ministry of Agriculture responsible; no data available	Not mentioned	Regional authorities responsible; no data available;	Obligatory bookkeeping including buffer zones kept;local environmental authorities responsible; punishment possible; farmers may leave financial support; monitoring data show that the approach is successfull	Records in a special form for LERAP obligatory; Not keeping the buffer zone is an offence but no data on enforceability
No spray or no crop zone	Not mentioned	2 m adjacent to water body no-cultivation zone	Not mentioned	Not mentioned	No spray zones	Not mentioned	Not mentioned	Not mentioned	Buffer zone must be no-crop area (or othe rcrop than on field)	Main Environ-mental legislatiom: 500 m adjacent to main dam reservoirs.	Not mentioned	No crop zone adviced to avoid plant protection problems	Not mentioned

	Austria	Denmark	Finland	France	Germany	Greece	Ireland	Italy	Netherlands	Portugal	Spain	Sweden	UK
Miscellan-eous	Not mentioned	Additional uncertrainty factor in use; 10-12 m buffer zones should be established adjacent to large waterbodie in any case without considering risk	1 and 3 m buffer zones in the frame of subsidies programmes, > 90 % of farmers participate; risk based mitigation measures planned	Not mentined	Not mentioned	Not mentioned	Quality standard may be mentioned in safety phrase waiting for filling of annex IV and V	Not mentioned	Certification system for farmers to increase transparency Arrangement between farmers and government as regards risk mitigation measures Special regulation for essential applications	Act under approval to regulate the application, trade and distribution of PPP, including certification and training of application operator and responsible technicians.	Not mentioned	Sugar beet farmers must sign contracts with the sugar industry which include the above mentioned environmental quality criteria;	Not mentioned
Training of farmers; book-keeping	Not mentioned	Not mentioned	Not mentionde	Well organized agricultural advisory system t125 Experimental and demonstrative catchments	Education of farmers and information campaigns mentioned	Not mentioned	Not mentioned	Not mentioned	License to spray;Training programme;Book keeping of used pesticides	Training for farmers and responsible technicians (Act under approval) Bookkeeping obligatory for farmers at IPM, and those supported by agro-environmental neasures	Not mentioned	Education of farmers and information campaigns	

1.2.1 How comparable are risk mitigation measures currently used in the EU Member States ?

The overview presented above clearly shows that the risk mitigation measures currently used to protect aquatic organisms – and it seems to be the best case - are very heterogenous. Whereas in some MS basically all water bodies are protected except those which fall dry over very long periods of the year others differentiate the protection level between man made/ small ditches and more natural ponds, lakes and streams. This is a very important issue because these ditches are in very large areas of Central Europe the most frequently occurring type of water bodies adjacent to fields. The distinction between these two types of water bodies is not risk based but rather a more general or practical approach. However, there is also some scientific evidence to do so (see section .5.1.4). The definition of the term water body or surface water is also very important as regards irrigation systems.

Spray drift is the most often considered exposure route and the FOCUS Drift Calculator (FOCUS 2003) is frequently used to determine the width of the buffer zones. PECs for distances of up to 50 m can be determined and those distances where no effects on aquatic organisms are to be expected can be used for setting risk mitigation measures. These drift values are supported by measured data in field trials. There are also other drift values of high quality available which are used in a few member states for setting risk mitigation measures.

Local conditions which mitigate risk are only considered in a few MS. Risk mitigation measures only related to the standard scenario for the risk assessment (standard application technique, 30 cm deep stagnant water body, well structured community) are protective but also based on an overestimated risk prediction in a high number of use situations. To avoid the latter one a few MS started to implement differentiated risk mitigation measures which take into account local use conditions like running water bodies, windbreaks, reduced application rate and especially spray drift reducing application technique. The data to decide upon the risk reducing potential of these conditions are not as good as for example those which form the base for the Ganzelmeier values. However, in any case it is possible to decide on the base of conservative estimates.

In France for example runoff is considered as the most important exposure route. On the regional level mitigation measures are set if monitoring results show that in a catchment environmental quality standards are breached. In other MS runoff is also considered when setting risk mitigation measures. In Germany for example grassed buffer zones with different width are used but also conservation tillage and detention ponds are regarded as appropriate tools.

Exposure via drainage systems is only possible where these systems still work. Regarding old systems it is often difficult to decide upon this. Furthermore, soil properties are important and risk mitigation measures for example in the UK restrict the use to relevant areas. An effective mitigation measure currently used is to restrict the period of use of the product to such month where exposure via drainage systems is not very likely.

The enforcement of risk mitigation measures is an important issue. However, there are only very limited data regarding the acceptance of restrictions by farmers and the control activities of MS available. The same is true for the methods used to enforce the restrictions. In Germany residues of plant protection products are measured within the buffer zones and in the middle of the field. If differences in residues are too low, an offence is considered to have occurred and punishment of the farmer may result. In case of herbicides it is usually possible to see whether the abundance of weeds in the buffer zone is comparable with the situation in the middle of the field. Most effective are systems where farmers are able to incorporate their buffer zones in set aside programmes like in Sweden. Furthermore it seems reasonable if special contracts with food industry or government require the farmer to follow environmental quality standards. It is very important to look whether other authorities also working in the rural area like for example those responsible for nature conservation, fertilizer use etc. set also restrictions or grant financial support for environmentally friendly activities of farmers. Cooperation might be possible and contradictions should be avoided in any case.

Training of farmers is a very important activity to make them familiar with risk mitigation measures. However, not too much detailed information is available on training programmes in MS. Only well informed farmers are able to understand why risk mitigation measures are set. Acceptance can be increased by training and information and this hopefully reduce efforts for controlling farmers. If available, more environmentally friendly products should be proposed.

1.3 Risk mitigation for spray drift

Spray drift is influenced by many factors, including those related to the outdoors environment and meteorological conditions, the spray technique and the crop and its canopy structure. Discrimination is made between relationships found for arable crop spraying and those for orchard spraying.

1.3.1 Arable crops

1.3.1.1 Factors determining spray drift deposition

Wind speed

Most spray drift experiments reported in the literature were not designed to measure the effect of wind speed. In cases where researchers mention the effect of wind speed, it is mostly interacted with nozzle type and sprayer boom height (Smith et al., 1982; Gilbert & Bell, 1988; Arvidsson, 1997). However, a strong positive correlation has been found between wind speed and spray drift deposition. Arvidsson (1997) found a change in total measured drift as soil deposit in the area 1-5m from the field edge and as airborne up to 6m height at 5m distance from 6% to 4%, being a 50% decrease in spray drift when wind speed (measured at 2m height) decreased from 4 m/s to 2 m/s.

Crop type

Ganzelmeier et al. (1995) found only minor differences in spray drift when spraying a cereal crop and a bare soil surface. Therefore only one set of drift values were proposed for field crops. Stallinga et al. (1999) measured the effect of height of a wheat crop on drift. It was found that there was no difference between the drift for 40 cm high summer wheat and 80 cm high winter wheat. For both crop heights, however, the drift was greater than for spraying on bare soil (Figure 1.1). Van de Zande et al. (2001) found a difference in spray drift because of crop type (Figure 1.1). A significant difference in spray drift occurred between crop types and especially between a crop and bare soil surface, being less for the bare soil surface.

Crop-free buffer zone

The distance between the edge of the crop and the bank of the waterway is essential in determining spray drift deposition on the water surface. In the period 1992-1994, Porskamp et al. (1995) assessed spray drift for field sprayers applying spray volumes of 150 and 300 l/ha, and using either a Fine or a Medium spray quality (Southcombe et al., 1997). Sprayer boom height was set to 0.7m above the canopy of the potato crop. Within this volume range, the spray quality did not significantly affect the drift deposition in the experiments. Spray drift deposition at a distance of 2.25-3.25 m from the last potato row was on average 5.3% for both nozzle types sprayed conventionally (Figure 1.1). Compared to the conventional spraying, a field boom sprayer with air assistance achieved a 50% reduction in spray drift on the soil surface at the same downwind distance. Increasing the distance from the last nozzle to the surface water by means of a non-cropped spray-free zone of 2.25m (3 potato ridges) reduced the deposition by 70% on the surface water (Porskamp et al., 1995). Spray drift deposition changed therefore from 5-6% to 2.5-2.6% at 2-3m distance.

Vegetation on buffer zone

It is recognised that the structure of both target crop and plants in the margin between the sprayed swath and water can have a large influence on rates of deposition to surface waters.

In a series of field experiments, Van de Zande et al. (2000) assessed spray drift when spraying a sugar beet crop. Next to the crop, the field margin was planted with a 1.25m wide strip of *Miscanthus* (Elephant grass) cut at different heights just before spraying. Heights varied between not planted (0m), at crop height (0.5m), 0.5m above crop height (being sprayer boom height, 1.0m) and 1m above crop height (1.5m). Spraying was performed with a conventional and an air-assisted sprayer. Spray volume was 300 l/ha using Medium spray quality nozzles. The height of the windbreak had a clear effect on spray drift deposit. Spray deposit at 3-4m distance from the last nozzle decreased significantly with increasing heights of the *Miscanthus*. When *Miscanthus* was cut to the same height as the sugar beet, spray drift reduction was 50% compared to spray drift at the same distance when no windbreak was grown. Spray drift was reduced by 80 and 90% with *Miscanthus* 0.5 and 1.0 m above crop height, respectively.

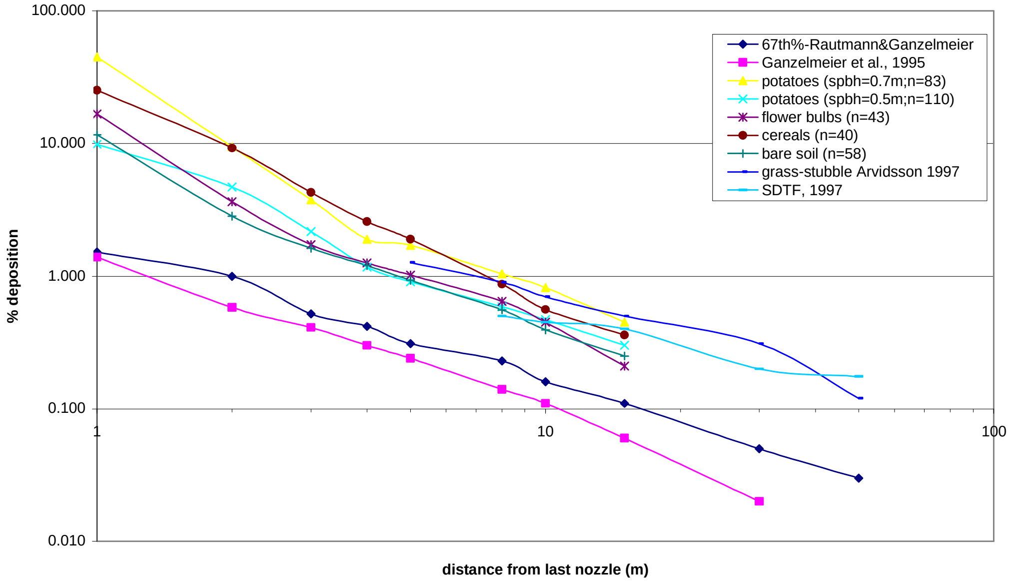
De Snoo (1995) found that the creation of a 3-m spray-free buffer zone in the field decreased drift deposition in the ditch by a minimum of 95%. With a 6-m no-spray buffer zone in the field alongside the waterway, no drift deposition in the ditch could be measured. Miller &

Lane (1999) present the results of wind tunnel experiments examining the distribution of airborne spray from simulated boom sprayer application systems simulating operation over bare ground or short crop conditions. Results from these measurements showed that the risk of drift with a grass and wild flower mixture compared with a 200 mm cut stubble was reduced by up to 34.7 %.

In a summary of observations from field studies prepared by Mackay et al. (2002) on behalf of UK Pesticides Safety Directorate, it is mentioned that in studies conducted by Taylor et al. (1999) a boom sprayer operating over a tall grass surface gave levels of drift in the range of 138% to 270% (1.0 – 2.0 m downwind of the sprayed swath) of those for an equivalent sprayer operating over a short grass surface. At greater distances (4.0 – 5.0 m downwind) the drift reduced to between 56% and 62% of the comparative short grass figures. The mitigation afforded by a margin comprised of grass and wild flower mixture with a base canopy height of 0.7 m with elements extending to 1.3 m high was of the order of 60 – 85% relative to drift observed with a 0.15 m mown grass margin (Miller et al., 2000).

Koch et al. (2002) indicate that the spray drift deposition on a field crop edge or boundary vegetation differs from deposition on the ground (soil surface). As spray drift consists mainly of droplets smaller than 100 μm and drift deposits are single droplet patterns retained on any surface, coverage defines the effect of spray drift on vegetation. Drifting particles are mainly retained in the upper zone of a canopy according to wind and air movement. Droplets barely penetrate into lower canopy regions. Drop distribution is very scattered and therefore the effect of spray drift on boundary vegetation is more variable than suggested from the figures of spray drift deposition measurements in drift trials. Koch et al. (2002) conclude that the dose response from spray application is different from the dose response from drift deposition. The smaller proportion of droplets <100 μm in the spray from low-drift nozzles has been shown to decrease desiccation from herbicide spraying in the drift area next to the sprayed field.

Figure 1.1. Data from various sources on the effect of crop type and environmental conditions on spray drift (values are 50th percentiles unless otherwise stated)



Ditch lay-out

Spray drift is usually measured on a bare soil surface next to the sprayed field. However, spray drift data are most often used for determination of ecotoxicological effects in the water of a ditch. Porskamp et al. (1995) found that when spray drift was measured just above the water surface it was around 30% lower than when measured at the same distance on a bare soil surface. Spray drift deposition changed therefore from 5-6% at 2-3m distance to 2.5-3.7%. Including a crop-free buffer zone of 2.25m, spray drift deposition on surface water in the ditch changed from 1.4-1.6% to 1.1-1.3%. Redistribution of the spray deposit takes place over the banks and the surface water area. The area of the banks is larger than that of the ground area on top of it. Also the change in air-flow pattern in the ditch compared to bare soil surface influences redistribution. In general it was found that in a 4m wide ditch the field-side bank spray drift deposition was 60%, on the surface water 70% and on the opposite-side bank was 104% of the spray drift deposition on ground level at the same distance.

Driving speed of sprayer

Arvidsson (1997) found a positive correlation between driving speed and spray drift. When driving speed was increased by 1m/s spray drift deposition was increased by 1.0%; within the trajectory of 1 m/s to 2.5 m/s velocity, this means a spray drift deposition of 4.2 and 5.8% respectively on the zone 1-5 m next to the field. Miller & Smith (1997) found an increase in airborne spray drift of 51% when forward speed was increased from 4 to 8 km/h and by 144% when the speed was further increased to 16 km/h.

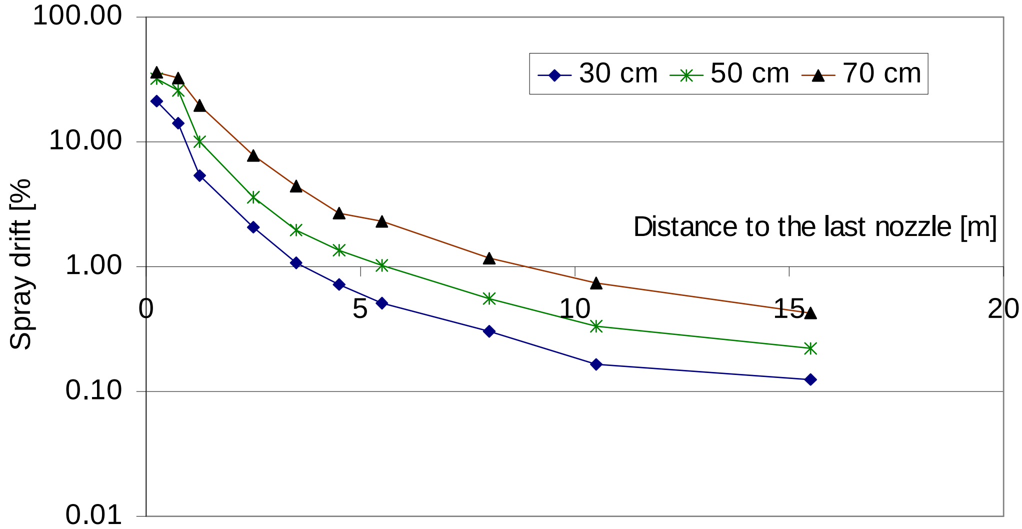
Sprayer boom height

Sprayer boom height is correlated with spray drift. With increased sprayer boom height spray drift deposition is also higher. De Jong et al. (2000a) undertook comparative drift measurements and found an effect of sprayer boom height (Figure 1.2). Spray drift was reduced by 56% when sprayer boom height was reduced from 0.7m to 0.5 m. The same reduction (54%) occurred when the sprayer boom was lowered from 0.5 to 0.3 m above crop canopy.

Although not compared in the same experiments but based on a number of replicate measurements, it can be concluded that a decrease in sprayer boom height from 0.7m (experiments 1992-1994) to 0.5m (experiments 1997-1998) above a 0.5m crop canopy reduces spray drift by 70% at a distance 2-3m from the last nozzle (Figure 1.1) when spraying a potato crop (300 l/ha). When sprayer boom height was reduced, the effect of air assistance on drift reduction increased from on average 50% for the 0.7m boom height to 70% for the 0.5m boom height (Van de Zande et al., 2000d).

Arvidsson (1997) found that spray drift deposition was reduced by 40 % when sprayer boom height decreased from 1.7 m to 0.9 m above the crop canopy. Arvidsson (1997) found that a further decrease in sprayer boom height to 0.6m and 0.3m above the crop canopy resulted in a spray drift reduction of 55 and 75%, respectively. Taylor et al. (1989) and Ripke (1990) reported similar effects.

Figure 1.2. Effect of sprayer boom height (30cm, 50cm and 70cm above crop canopy) on spray drift deposition next to the field when spraying a potato field (spray volume 300 l/ha, Nozzle XR11004 @ 3bar; de Jong et al., 2000a).



Sprayer boom height is commonly set to a height of 0.5m above crop canopy. The sprayer boom generally moves between 0.9m and 0.3m in height because of in-field bouncing. Spray drift variation because of this movement has been estimated to be between +/- 40% (Arvidsson, 1997) and +/- 55% (De Jong et al., 2000b).

<u>Spray volume, nozzle type and air assistance</u>

Depending on the type, the size and the pressure used, nozzles produce a spray with drops of different sizes. The distribution of drops in a spray fan can be measured and classified to spray quality classes (Doble et al., 1985; Southcombe et al., 1997). Different amounts of spray drift are produced because of these differences in spray qualities, the speed of the drops in the fan, and the spray volume distribution in the fan (top angle of the nozzle). The influence of nozzle type on spray drift depends on interactions with sprayer boom height, wind speed, and pressure (Elliot & Wilson, 1983; Gilbert & Bell, 1988; Ripke, 1990, Ripke & Warnecke-Busch, 1992). In general it can be said that the coarser the spray quality the lower the spray drift. This can also be found in recent studies on the effect of low-drift nozzles on spray drift (Van de Zande et al., 2000a; Van de Zande et al., 2000b; Ganzelmeier & Rautmann, 2000; Herbst & Ganzelmeier, 2000; Walklate et al., 2000, Hewitt, 2000). Spray drift reductions up to 85% when spraying at 300 l/ha with a field sprayer equipped with Very

Coarse nozzle types are possible compared to a standard flat fan nozzle Medium Spray Quality.

IMAG performed field tests on spray drift in 1997, 1998 and 1999 to quantify the effect of two spray volumes using “low-drift” nozzle types and air assistance (Van de Zande et al., 2000b). Spray drift was quantified for a series of low-drift nozzle types all applying a spray volume of 150 l/ha and 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 (XR), drift guard (DG), anvil flat fan (TT) and two types of injection nozzles (ID and XLTD) were evaluated in the field. All nozzles were used in a conventional way and with the use of air assistance (Hardi Twin, full capacity - nozzles kept vertical). The height of the potato crop canopy was 0.5 m. Results (Figure 1.3 and Figure 1.4) show that the terminology “low drift nozzle” needs further specification. From the experiments it became clear that within the group of low drift nozzles a ranking by level of drift reduction is preferable. The comparison with a standard sprayer-nozzle configuration is of value, also for comparison of the results with other drift experiments.

Although a spray volume of either 150 l/ha or 300 l/ha was used with all nozzles, the difference in the range of droplet sizes resulted in drift reductions up to more than 85% when compared to a XR11004 nozzle (Van de Zande et al., 2000b). The terminology ‘low-drift’ nozzle therefore needs further specification.

Injector nozzles used on field sprayers in Germany resulted in spray drift reduction of 50-90% compared to the basic German drift values (Schmidt, 2001). This was said to be relatively low because the existing drift values for field crops are already very low. Spray drift reduction could be increased when the injector nozzles are used in special application, i.e. nozzle size larger than 04, pressure of 2-3 bar, driving speed 5 km/h or lower, resulting in a spray volume of at least 300 l/ha.

Finally it is important to recognise that in several crops – especially horticultural - no drift reducing technique is available. In strawberries, for example, the standard values are much lower than the usual drift values for field crops.

Figure 1.3. Relative spray drift deposit at 2-3m from the last nozzle for different low-drift nozzles (ISO 04 @ 300 kPa) and air assistance (+A) when spraying potatoes with a spray volume of 300 litres ha^{-1}. Standard nozzle type is XR11004 (=100). (Van de Zande et al., 2000a)

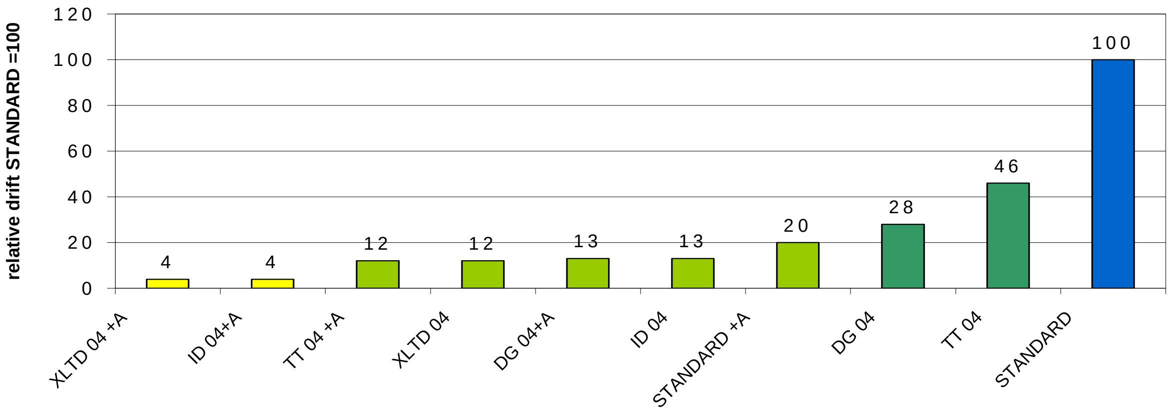
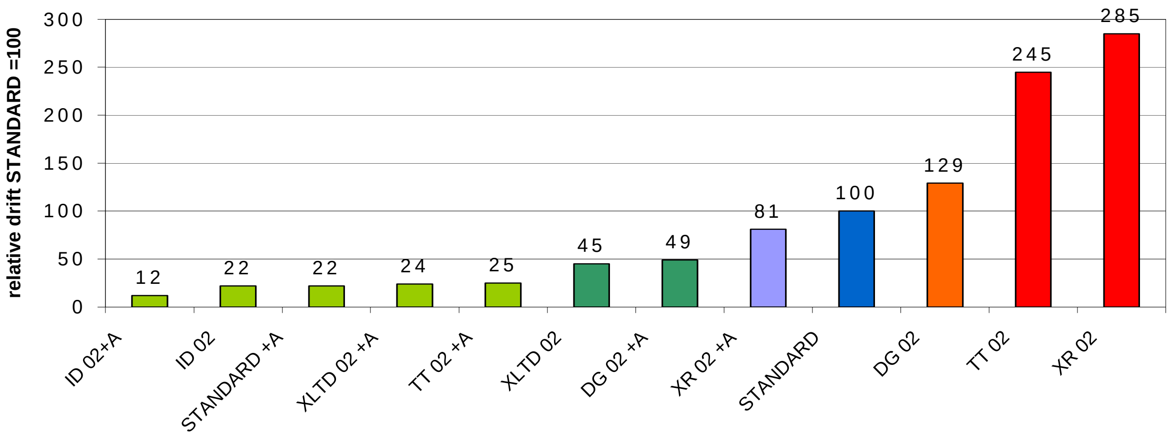


Figure 1.4. Relative spray drift deposit at 2-3m from the last nozzle for different low-drift nozzles (ISO 02 @ 300 kPa) and air assistance (+A) when spraying potatoes with a spray volume of 150 litres ha^{-1}. Standard nozzle type is XR11004 (=100). (Van de Zande et al., 2002a)



Taylor et al. (1999) showed that spray quality dominated over operating pressure and wind speed for conventional nozzles, while drift-reducing pre-orifice and air-inducing nozzles reduced drift losses by more than 75% at equivalent outputs. Air assistance in combination with these drift reducing nozzles reduced drift fallout by up to 95% compared to a conventional Fl10/1.6/3.0 nozzle without air assistance.

Air assisted spraying in general reduced spray drift by more than 50% (Taylor et al., 1989; Ripke, 1990; May, 1991; Ringel, 1991; May&Hilton, 1992, Porskamp et al., 1995, Porskamp et al., 1997; Schmidt, 2001). In combination with low-drift nozzles and a sprayer boom height of 0.5m above crop canopy, Van de Zande et al. (2000a, 2002a) found an average spray drift reduction because of the use of air assistance of 70%.

Tank additives

An alternative approach to nozzle modification is to introduce a drift control additive to the spray mix, designed to increase droplet size. Such additives are common in the US and Australia, where higher ground speeds at application are typical. A number of oil or synthetic latex-based products are available for use in the UK. However, problems can arise if sprays become too coarse, resulting in reduced retention and uptake. Research has demonstrated a significant (20-50%) reduction in drift when using chlorpyriphos with the addition of a synthetic latex anti-drift agent (Thacker et al. 1994; Mackay et al, 2002)

Band spraying

The drift caused by the use of a band sprayer was recorded during field measurements. The sprayings were carried out in sugar beet and maize crops with row spacing of 50 cm and 75 cm, respectively. The band-sprayer was equipped with either one or two nozzles per row of respectively a Medium or a Fine spray quality. Spray volume for the band sprayer was 130 l/ha and 200 l/ha for the maize and the sugar beet crop respectively, defined by the difference in row width of both crops (0.75 and 0.50m respectively). Crop height of the sugar beet (4-8 leaves) and of the maize (3-5 leaves) was 10-15 cm.

The drift reduction due to the use of the band sprayer was 90% compared with a field sprayer (300 l/ha, medium nozzle type). The drift reduction was achieved both with a single-nozzle and a dual-nozzle version per crop row (Van de Zande et al., 2000c). These findings are supported by data from Germany (Rautmann, personal communication).

End nozzle

Overspray of plant protection products when spraying the edge of the field can be reduced by the use of an end-nozzle. An end nozzle produces a cut-off spray fan like that 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 0.2m to the outside (potatoes). An end nozzle (UB8504) in combination with a low drift nozzle (DG11004) reduced spray drift by 20% (60% with air assistance) at 2-3m distance from the last nozzle (Van de Zande et al., 2000b). At 1-2 m distance this effect was 50% (80% with air assistance).

Shielding

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 for a shielded field-sprayer and a prototype tunnel sprayer for bed-grown crops (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, lilies or a flower-bulb look-alike crop, cut mustard. No effect of these crop types was found on spray drift data. Also no effect on spray drift was found from the nozzle type used. A shielded sprayer boom reduced spray drift deposition at 2-3m distance from the last nozzle by 50%. A tunnel sprayer for bed-grown crops reduced spray drift by 90%. Boom coverage on a field sprayer as developed in the USA was said to be unsuitable for German conditions (Schmidt, 2001). An adapted system with a special folding technique of the boom for transportation on the road resulted in a 50% drift reduction without changing nozzle type.

1.3.1.2 Assessment of observed differences in spray drift deposition

Experimental design

Rautmann (personal communication) reported that the German spray drift data were gathered from experiments performed by research institutes and agrochemical manufacturers. The data are based on the following parameters:

number of experiments	50
windspeed	1 - 4.9 m/s
temperature	6.7 - 24.45 ^{o}C
driving speed	6 km/h
spray pressure	2.1 - 5 bar
spray volume	150 - 300 l/ha
measuring distances	1, 5, 10, 15, 20, 30, 40, 50, 75 and 100m
nozzle types	4110-20, 11004, DG11002, DG11003, DG11004, ID12002, ID12003, ID120015, TD025, XR11002, XR11003, XR11004VS, XR11004

The spray drift experiments in the Netherlands have a comparative nature (Van de Zande, 2001). Spray techniques are compared by spraying a cropped area and measuring spray drift on a bare soil surface next to the cropped area. Differences are statistically evaluated based on replicate measurements (at least 10) in time. A reference spraying system was always taken into the experimental set-up, being a standard field sprayer using standard flat fan nozzles.

The Spray Drift Task Force (SDTF, 1997a) compared effects of spray techniques with a reference sprayer. Comparative measurements were always performed in combination with the reference spraying system. Measurements were performed on short grass.

Measurements in Sweden were performed on short grass (Arvidsson, 1997). A reference spraying system was used to make comparisons of spray technique, sprayer boom height and nozzle type. Differences in spray drift between parameters/objects were statistically analysed.

Crop type

From the database on spray drift it was concluded that all measurements were performed either in the spring or in the autumn. The remarks on crop stage mention winter barley, arable land, and field for the early measurements (1990). These data were based on approximately 25cm height winter cereal and bare soil surface. The additional experiments of the expanded database contain measurements on either short cut grass/cereal stubble or bare soil surface. In fact all measurements are therefore based on bare soil surface/short cut vegetation. Data from Van de Zande et al. (2001) show that spray drift deposition when spraying a bare soil surface results in a lower drift value for a given distance.

Nozzle type

The initial database (Ganzelmeier et al., 1995) was based on eight experiments performed in 1990 with Medium (Doble et al., 1995; Southcombe et al, 1997) Spray Quality nozzle types. Apart from these standard flat fan nozzles, the expanded database contains 23 experiments with comparable flat fan nozzles which are commonly used in arable field spraying but also DG-type (8 experiments), ID-type (7 experiments) and TD-type nozzles (4 experiments). These nozzle types are advertised and sold as low drift nozzles. Depending on size and pressure this will also be the case as is shown in field research (Van de Zande et al., 2000), modelling (Porskamp et al, 1999) and windtunnel experiments (Walklate et al., 2000). Some of the mentioned nozzles have a spray drift reduction status in the UK (LERAP low drift star rating), as compared to a standard (11003; F110/1.2/3.0) flat fan nozzle.

The German drift deposition database represents a special deposition situations in so far, as:

- 19 out of 50 experiments are based on measurements with a potential "low-drift" nozzle type spraying. However, even when the data from the 19 experiments would be removed from the data base the drift values would not change considerably because the 90th percentiles are used;
- They concern situations with a bare soil surface or short cut canopy.

The Dutch database reflects higher spray drift depositions because all measurements were performed with a Medium spray quality nozzle. A further analysis of the Dutch, German, US and Canadian database on spray drift is underway.

1.3.2 Orchards

1.3.2.1 Factors determining spray drift deposition

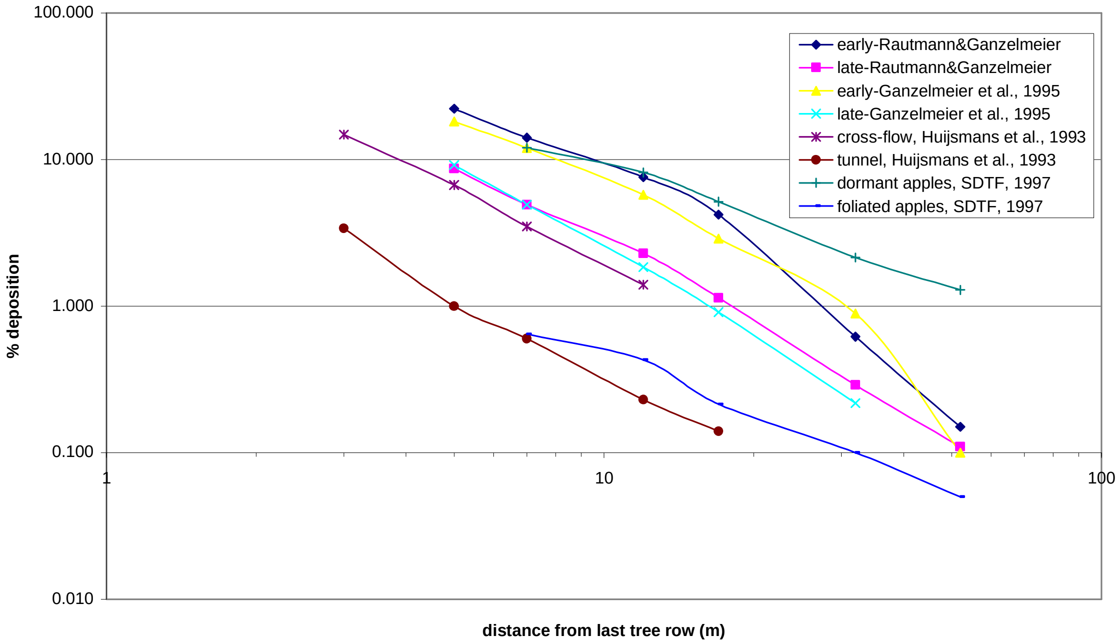
<u>Wind speed</u>

The effect of wind speed on spray drift in orchard spraying (Ganzelmeier et al., 1995; SDTF, 1997) was only found when dormant (no foliage). A change in wind speed outside the orchard was only reflected by a change in wind speed inside the orchard when the trees were bare (SDTF, 1997).

<u>Foliage density</u>

Spray drift deposition was approximately 22 times greater (Figure 1.5) at 7.5m distance (from the field) from dormant compared to foliated apples (SDTF, 1997). Ganzelmeier et al. (1995) found on average 2-3 times higher spray drift deposition at 5-7m distance (Figure 1.5) from the last tree-row when spraying dormant instead of foliated apple trees (resp. 12.0% and 4.9% drift deposition).

Figure 1.5. 50th percentile spray drift values in orchard spraying - effect of foliage density (early+dormant, late+foliated) and sprayer types (cross-flow, tunnel)



<u>Windbreak</u>

A windbreak of alder trees on the outer-edge of the field (Figure 1.6; bottom) reduced spray-drift by 70-90 % in the zone 0-3 m downwind of the windbreak (Porskamp et al., 1994a). Walklate (2001) measured passing spray cloud in front and behind a 7m high row of alder trees. A single avenue of trees was sprayed in the orchard. In the early season, an open structure resulted in a similar distribution on both sides of the windbreak. In a full leaf canopy situation, the spray cloud was moved upwards behind the windbreak to have a maximum at 7.5m height. Typical reductions of 86-91% for a 7m alder tree were found. Richardson et al. (2002) found drift reduction from an alder tree hedgerow of 50% when in full leaf. Large differences do occur between the measured effects of windbreaks on spray drift reduction (Ucar & Hall, 2001) especially because of geometrical construction of the leaf canopy leading to differences in capture efficiency of passing droplets and redirection of the wind profile around the windbreak. Research of Wenneker et al. (2003) shows the effect of leaf density of an alder tree windbreak on drift reduction. A bare windbreak resulted in a drift reduction of 20% measured at 3m distance behind the trees (resembling the stem and branches area). When leaves start to develop, drift reduction increases to the values found by Porskamp et al. (1994). Large differences do however occur between species of windbreak trees. Canopy density varies between leaf trees such as alder, poplar and willow (Wenneker et al., 2003) but also between needle-like foliage, which captures two to four times more spray than broad-leaves (Ucar et al., 2003).

Emission shield

An emission shield (gauze 40% permeability) on the edge of the field and of equal height as the fruit trees (2.5 m) reduced spray drift in a full leaf orchard by 60% (Zande et al., 2001).

Single sided spraying of outer tree rows

The effect of single-sided spraying of the outer tree row, an emission shield on the edge of the field and the growing of reeds in the ditch on drift reduction were reported by Van de Zande et al., 2001. Spray drift reductions of 45% were found for the single sided spraying of the outside tree row. A similar approach is used in Germany (Schmidt, 2001). A drift reduction of 75-85% can be reached when the last five rows at the edge of the orchard are sprayed without air assistance in the wind and drift direction and coarse nozzles are used with the nozzles on the downwind side mounted on a vertical boom.

Sprayer type

Spray drift deposition is influenced by sprayer type. When the spray is more directed towards the tree canopy, as with a wrap-around sprayer, spray drift deposition from the last tree was reduced (SDTF, 1997), compared to an airblast sprayer (axial-fan).

The reference situation for orchard spraying in the Netherlands is a cross-flow fan sprayer (Figure 1.6; top) spraying in an orchard with leaves on the trees (LAI 1.5-2) and an average windspeed of 3 m/s. Spray drift for this situation and for drift reducing spray techniques such as a cross-flow orchard sprayer with reflection shields and a tunnelsprayer (Figure 1.6; middle) was assessed in the period 1991-1994. For the cross-flow orchard sprayer, the spray-drift deposition on the soil at 4.5-5.5 m downwind of the last tree was 6.8% of the application rate per surface area (Figure 1.5).

Compared to this reference situation, a tunnel sprayer achieved 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 increased spray drift 2 to 3 times compared to spraying trees with full foliage. In Germany it is found that a cross-flow sprayer has a 10-15% lower drift than an axial-fan sprayer (Schmidt, 2001). Recycling sprayers with a tunnel in orchards are reported by Schmidt (2001) to have a drift reduction of 90%.

Spray drift measurements carried out with conventional air-assisted axial fan orchard sprayers and vertical deflector sprayers in Spanish apple orchards showed a 50% reduction of spray drift at full development growth stage (Fillat et al., 1993).

Leaf sensor equipped sprayer

Koch & Weisser (2000) showed a reduction in spray drift of 50% by using a sprayer equipped with gap-detection sensors. A 50% drift reduction for sensor controlled sprayers was also reported by Schmidt (2001). These sensors prevent spraying the gaps between the top of the trees where no foliage is apparent. The effect of a sensor-equipped cross-flow sprayer on drift reduction was compared with a standard cross-flow sprayer equipped with the same nozzle-types (Zande et al., 2001). The drift reduction achieved with the sensor equipped orchard spraying was on average 22% and 50% for the no-leaf and full canopy situation respectively. Drift reduction depends very much on canopy structure. The overall reduction of pesticide use for the whole field is most important.

Air assistance

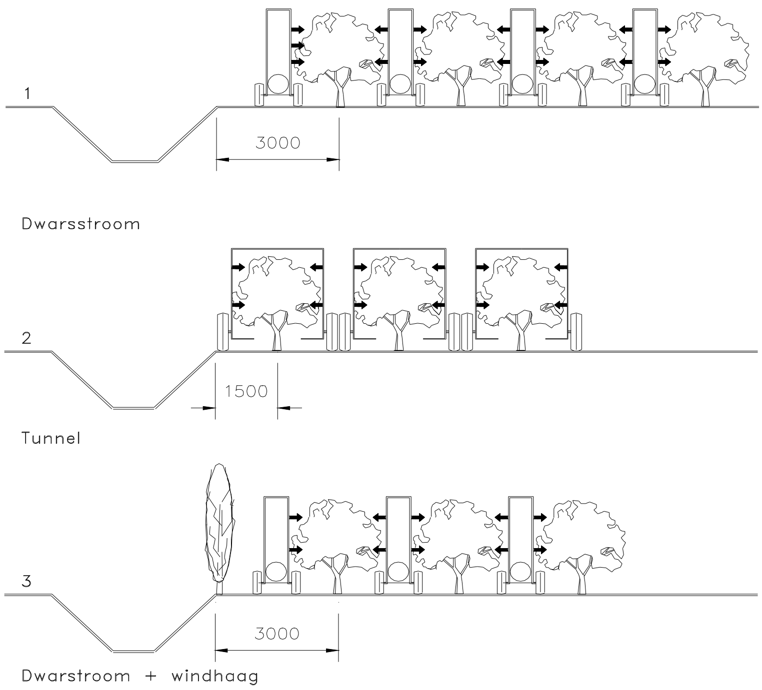
Sprayers in fruit growing are usually equipped with air assistance, predominantly to transport the spray from the sprayer towards and into tree canopy at higher heights, and to increase spray deposit on the target because of increased foliage movement. Cross et al. (2003) showed that the volumetric air flow rate of axial fan orchard sprayers influenced spray drift. Reducing volumetric air flow rate from 11.3 to 7.5 and 4.1 m^3/s reduced spray drift by <50% for the medium flow rate to 55-93% for the low flow rate. Drift reductions were however lower in stronger wind conditions especially early in the season when the canopy was less dense.

Lower spray drift amounts with decreasing air flow rates were also reported by Solanelles et al. (1997) and Walklate et al. (1996).

Sprayer speed

Solanelles et al. (1997) and Walklate et al. (1996) showed that increasing sprayer speed reduced spray drift.

Figure 1.6. Spraying systems and situations in orchard spraying (after Huijsmans et al., 1997).
Top: cross-flow sprayer spraying last tree row towards the field
Middle: tunnel sprayer
Bottom: cross-flow sprayer with a hedgerow planted on the edge of the field



<u>Nozzle type</u>

Heijne et al., 2002 found no effect of hollow cone venturi type nozzles on spray drift reduction in the short range from the last tree row (<8 m). They found a drift reduction of 65% for larger distances (10 m from the last tree row).

1.3.2.2 Assessment of observed differences in spray drift deposition

There is reasonable agreement on the height of the German, US and Dutch drift curves for early and late orchard spraying. Data from Ganzelmeier & Rautmann (2000) are in agreement (Figure 1.5) with data from other studies (Huijsmans et al., 1993; SDTF, 1997).

1.3.3 Bush trees

Doruchowski et al. (1999) found that when spraying blackcurrants, a directed air-jet sprayer produced lower spray-drift compared to a conventional sprayer. A superior penetration of bushes was observed despite 50% lower air volume produced. The loss to the air (recorded on a frame placed behind the bushes) produced by the directed air-jet sprayer (SEPIA) was several times lower (95% drift reduction) than that of the conventional sprayer.

1.3.4 Nursery trees

In a series of experiments (1996-1997) in high nursery (alley) trees, a conventional sprayer equipped with flat-fan nozzles was compared with a conventional axial fan sprayer with hollow cone nozzles (Porskamp et al., 1999a). The comparison was made for two tree types: spindle form and transplanted alley-trees. The level of spray drift deposition next to the sprayed field did not differ for the two nozzle types. The spray drift deposition on the soil at 3-4 m from the last tree row was 13.6% for the transplanted trees and 3.3% for the spindle trees.

1.3.5 Vineyards

Spray drift reduction in vineyard spraying in Germany is mainly achieved with sensor-controlled sprayers and tunnel sprayers (Schmidt, 2001). Drift reductions reported are 50% for the sensor-equipped sprayer which can be increased to 90% when used in combination with very coarse spray qualities. A tunnel sprayer in vineyards can provide drift reductions of more than 90%. The same amount of drift reduction was also achieved where four rows adjacent to water bodies were sprayed without pressure in the direction of the water body (75% reduction for two rows). Planas et al. (2001) reports no spray drift losses (< 1% from sprayer output) from the application of tunnel sprayers in vineyards in Spain. The standard application with an axial fan orchard sprayer produced an airborne drift amount of 5-7% of sprayer output (300-360 l/ha).

1.3.6 Hops

Drift reduction can be 90% when hops are sprayed from outside with a partly covered fan in combination with injector nozzles (Schmidt, 2001). A tunnel sprayer adapted for the high growing hops also resulted in a 90% drift reduction (Schmidt, 2001).

1.3.7 Citrus

Spray application in citrus groves requires high volumetric air flow rates (more than 50.000 m3/h) to get enough spray deposit in the centre of the canopy. Measured spray drift with standard axial fan orchard sprayers is sometimes as high as 15% of the total sprayer output (Planas et al., 1998). A cross-flow tower sprayer reduced spray drift by 50% of the amount compared to a standard axial fan sprayer spraying orange trees.

1.3.8 Physical and chemical modifications of spray liquid to affect spray drift.

During a spray application the occurrence of spray drift may be a significant loss of spray liquid to downwind areas. Small drops may remain airborne long enough to evaporate considerably before impacting onto the ground. This evaporation comprises that of the solvent (usually water) and that of the solute (the pesticide and various adjuvants).

Several methods can be used to suppress spray drift during application (Ruiter et al., 2003). One may distinguish three options:

(a) diminish the production of small drift-prone droplets at the nozzle outlet;

(b) promote a rapid deposition of the drops;

(c) suppress the evaporation of drops in air.

The use of nozzles with a relatively coarse drop size distribution is one possibility to achieve option (a). Conventional nozzles with a coarse spray often imply relatively high dose rates. So-called ‘drift reducing’ or ‘low-drift’ nozzles may produce a relatively coarse spray, yet do not increase the applied dose. The draw-back is that initial velocity of the drops for these nozzles is relatively low, which partly undoes the advantage of the coarser spray. Other types of nozzle producing drops with a narrower size distribution clearly have an advantage, yet the commonly used hydraulic nozzles all have a similar (wide) ‘span’ of drop sizes.

To decrease the time that drops need to travel from nozzle to target (option (b)), the use of air-assistance to guide drops downward is a well-known but costly possibility. Charging drops is another option to promote deposition, yet in practice this is only possible for fine sprays. Besides, it can have the opposite effect as charged drops repel each other and a cloud of drops will tend to spread out.

Another option to diminish spray drift involves a ‘non-technical’ approach, the use of adjuvants to modify the physicochemical properties of the spray liquid. Roughly, two groups of adjuvants can be distinguished. The first group coarsens the drop size spectrum (i.e. option (a)), the second group suppresses evaporation (i.e. option (c)). In principle, deposition can be promoted by increasing the mass density of the spray liquid (option (b)). This is not a feasible option with adjuvants, yet using a ‘heavy’ solvent may occasionally have some advantages.

1.3.8.1 Reduction of in-flight vaporization

The evaporation rate of additives is determined largely by their saturated vapour pressure, which is much lower than that of water. The effect of a lower diffusion coefficient and a higher density is roughly compensated by the increase in molecular weight. The net result is an evaporation rate for additives which is much lower than that of water, even for additives known as ‘volatile’. Therefore, evaporation of water occurs much more rapidly for aqueous sprays than that of the suspended ingredients. Water will evaporate until a more or less dry particle remains. This particle may drift a considerable distance without a notable change in size.

Non-aqueous solvents may evaporate with similar speed as water (e.g. diethylbenzene, n-decane) or even faster (e.g. xylene).

A possible way to reduce drift is to prevent the drops from shrinking, i.e. prevent the solvent from evaporating. Though a non-volatile surfactant may act as an ‘evaporation retardant’ by shielding the droplet, only few authors investigated this aspect. Hall et al. (1994) describe an apparatus to determine the evaporation rate of a relatively large pendant drop in a static environment. They found that several adjuvants appear to reduce evaporation. Unfortunately, a possibly drift reducing effect under practical conditions cannot be deduced from such an experimental method. Besides, an evaporation-retardant adjuvant does not *prevent* drift occurring, it merely reduces the distance traveled by drift-prone drops. Probably more importantly, the use of such non-volatile surfactants affects the drop size distribution, which is a more effective way to influence spray drift.

On the other hand, vapour loss of active ingredients in air is more likely to originate from spray deposits than from evaporating airborne drops (Hartley and Graham-Bryce, 1980). Only the smallest residual particles (after complete evaporation of the solvent) that may stay airborne over long distances may contribute significantly to vapour loss. Using the spray drift model IDEFICS (Holterman et al., 1997) it has been estimated that roughly 3% of the spray liquid is lost into air (as vapour and 'dry' particles) during a conventional application on field crops (Smidt, 2000).

1.3.8.2 Increasing drop size

Physical properties of the spray liquid may affect the drop size distribution of a spray. Spray coarsening by physicochemical modification is thought to result from early break-up of the liquid sheet that forms at the nozzle outlet. In that case the liquid sheet is still relatively thick, and this is directly related to the sizes of the drops produced.

The most relevant physical properties to affect the moment of break-up are viscosity and surface tension. Homogeneity of the bulk liquid, particularly in the case of suspensions and emulsions, can be an important factor as well. The following sections describe the effect of these properties on drop size distribution.

<u>Viscosity</u>

Adjuvants that increase viscosity of the spray liquid coarsen the spectrum of the spray (Moser and Schmidt, 1983). Dynamic viscosity (η) of pure water is about 1.0 mPa·s (at 20^{o}C). A liquid with dynamic viscosity of about 1.6 mPa·s significantly affects drop size distribution (Moser and Schmidt, 1983). Though many additives hardly affect viscosity, some may affect viscosity considerably. Occasionally, even a relatively dilute 0.1% solution of the proper adjuvant may increase dynamic viscosity up to 1.9 mPa·s, and thus decrease spray drift due to coarsening of the spray (Moser and Schmidt, 1983).

Viscosity is usually temperature dependent. Even for a 'simple' fluid like water, a change in temperature may affect drop size distribution through a change in viscosity. Table 1.2 shows values of the dynamic viscosity for water at various temperatures (Margenau et al., 1953).

Table 1.2. Dynamic viscosity of water as a function of liquid temperature

Temperature (^{o}C)	Dynamic viscosity [1] (mPa·s)	Relative change in SMD [2]
0	1.79	1.09
20	1.01	1.00
40	0.66	0.94
60	0.47	0.89

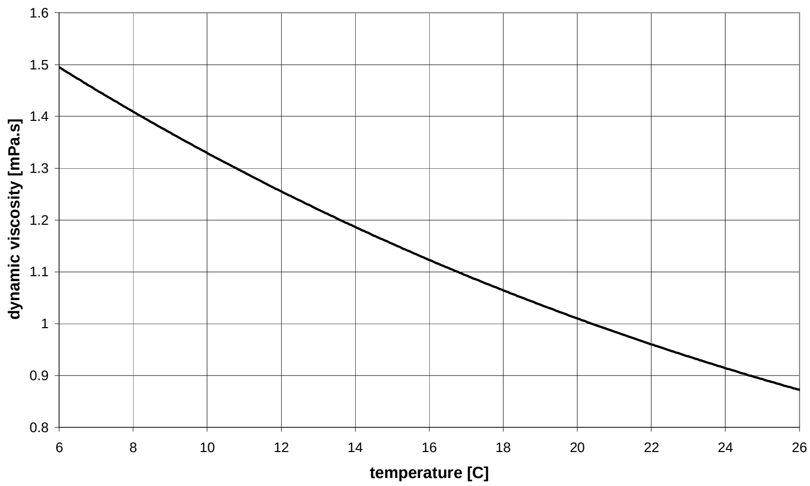
[1] Margenau et al., 1953

[2] Taking SMD (Sauter Mean Diameter) at 20^{o}C as a reference; see text.

By interpolation, one may estimate viscosity of water at intermediate temperatures. Figure 1.7 is obtained in this way. From this curve it can be estimated that at 20^{o}C the rate of change in viscosity of water is about -0.026 mPa·s per ^{o}C.

Often the Sauter Mean Diameter (SMD) is expressed as being proportional to η^{b}. Experimentally obtained values for b range from 0.06 up to 0.22 (Lefebvre, 1989). The parameter b appears to be related to flow rate: an increased flow rate will decrease b (Lefebvre, 1989). Using an average value of $b = 0.15$, SMD for a water spray changes only slightly for practical temperatures (see Table 1.2). Even when cooling the water from 20^{o}C down to (just above) zero, on average only a 9% increase in mean drop size is to be expected.

Figure 1.7. Dynamic viscosity of water as a function of temperature. Obtained by interpolation using the values of Table 1.2.



Aqueous solutions often are Newtonian liquids, i.e. their dynamic viscosity is independent of shear rate. However, adjuvants that aim to reduce drift by an increase in viscosity behave like non-Newtonian liquids: dynamic viscosity of the solution is a function of shear rate. The shear rate at the nozzle outlet is important for drop formation. Typically, shear rates at the nozzle outlet range from 10^4-$2 \cdot 10^5$ s^{-1} (Butler et al., 1969). For those solutions, viscosity decreases with increasing shear rate (‘shear-thinning’ behaviour). Therefore, to obtain a dynamic viscosity at the outlet of the nozzle that is significantly higher than that of water, the (low-shear) viscosity of the tank liquid must be even higher still (note that shear rates in the tank are low: typically 50 s^{-1}). Highly viscous liquids however cause problems with stirring and therefore with creating a homogeneous spraying liquid (Hartley and Graham-Bryce, 1980).

Non-Newtonian liquids have both viscous and elastic properties. Non-Newtonian behaviour is often accompanied by other (time-dependent) behaviour. Thixotropic liquids are shear-thinning when stirred, and the low-shear viscosity recovers only slowly after stirring has stopped. A high viscosity can lead to long threads in the formation of drops. These threads may break up into a number of small satellites. In the case of a highly elastic liquid, the thread and drop will recoil and stick together again after rupture, without the formation of satellites (Hartley and Graham-Bryce 1980).

For non-Newtonian liquids, Hewitt et al. (2001) distinguish extensional and shear viscosity. Extensional viscosity is particularly important when polymers with high molecular weight (typically $>10^6$) are used in the formulation. Even very low concentrations (~100 ppm) of such a polymer can lead to a significant increase in averaged drop size. The effect of extensional viscosity is related to the flow pattern through the nozzle, so that the effect on drop size may differ for different nozzles.

Surface tension

Another important physical factor is surface tension (σ) of the liquid. A lower surface tension will decrease average drop size. Several investigations state that SMD is proportional to σ^{a}, where a is about 0.25 (Lefebvre, 1989). Surface tension of pure water is 72.8 $mN \cdot m^{-1}$ (at $20^{o}C$; Moore, 1978). Suspending or solving various additives can easily decrease surface tension to about 30 $mN \cdot m^{-1}$, which causes a decrease in SMD of approximately 20%.

Aqueous solutions can both increase and decrease surface tension. Substances like fatty acids, whose molecules have both a polar (hydrophilic) group and a non-polar (hydrophobic) group, decrease surface tension when dissolved in water (Moore, 1978). The rate of decrease with concentration depends on the relative size of the hydrophobic group: the larger the non-polar group, the more pronounced the effect will be. Ionic solutions (salts) increase surface tension slightly, because the salty ions tend to pull water molecules into the interior of the liquid (by ion-dipole interaction), away from the liquid surface.

It should be noted that dynamic surface tension may differ from equilibrium (or static) surface tension, because obviously it takes some time for the molecules in the solution to form an equilibrium surface (Moore, 1978). If migration of molecules to the surface is very fast (typically $<10^{-3}$ s) static surface tension is applicable (Schmidt, 1980).

Surface tension of water is only slightly dependent on temperature, decreasing about 0.14 $mN \cdot m^{-1}$ per degree increase, averaged between 0^oC and 20^oC (Margenau et al., 1953). This means that in the practical range of temperatures, mean drop size of sprays of pure water is hardly affected by changes in temperature, as far as surface tension is concerned.

Tracers (dyes) dissolved in water are often used in experiments. Although these tracers do not affect viscosity or density significantly, the surface tension is usually lowered (typically to 60-65 $mN \cdot m^{-1}$ for some common fluorescent dyes at 1g/l; according to Schmidt, 1980).

Homogeneity of the bulk liquid

Dexter (2001) suggested that with emulsions the hydrophobic droplets, when they are stretched in the liquid sheet, form weak spots which promote early break-up of the liquid sheet and thus result in coarsening of the spray. There may be an effect of initial size of the emulsion droplets in the bulk liquid as well.

In general, dispersions do not appear to affect drop size distribution (apart from the effect of the physical properties described in previous sections), while emulsions show an increase in average drop size due to early break-up caused by hole formation, as well as a narrower size distribution (Hewitt et al, 2001).

Adjuvants for coarsening drop size spectrum

Tank Mix Additives

An alternative approach to nozzle modification is to introduce a drift control additive to the spray mix, designed to increase droplet size. Such additives are common in the US and Australia, where higher ground speeds at application are typical. A number of oil or synthetic latex-based products are available for use in the UK. However, problems can arise if sprays become too coarse, resulting in reduced retention and uptake. Research has demonstrated a significant (20-50%) reduction in drift when using chlorpyriphos with the addition of a synthetic latex anti-drift agent (Thacker et al. 1994; Mackay et al, 2002)

If tank mix additives are added to formulations they do not necessarily lead to the same results as for another product. In some products they might be effective in others not. This means they have to be tested in combination.

Butler et al. (1969) investigated several additives (Vistik 6.5 g/l: a hydroxyethyl cellulose; Dacagin 6.5 g/l: a polysaccharide; Norbak 7.5 g/l: a water-swellable polymer) that increased viscosity. The amount of small drops did indeed decrease. However, the drop size spectrum only shifted to larger drops, and the amount of large drops consequently increased. Due to non-Newtonian (shear-thinning) behaviour, relatively high concentrations had to be used to ensure a viscosity that was significantly higher than that of water at shear rates occurring in the outlet of the nozzle.

Schmidt (1980) tried solutions of Nalco-625 (a polyacrylamid-based emulsion) in water. Even a solution of 0.1 g/l significantly increased VMD (about 20%). The corresponding viscosity (1.27 mPa·s) of this non-Newtonian liquid was measured using a falling-ball viscometer, which should only be used for Newtonian liquids, and therefore the resulting viscosity is merely indicative. At the outlet of the nozzle the viscosity probably will be much closer to that of water. The reported surface tension was 32.9 $mN \cdot m^{-1}$. This means that viscosity and surface tension can hardly account for the observed differences in drop size spectrum.

Bouse et al. (1990) investigated the effect of several herbicide spray mixtures and polymer adjuvants on drop size spectrum. Although they found significant differences in VMD for herbicide mixtures without polymer adjuvants, variations in surface tension and viscosity were only small and could not explain the differences in drop sizes. Subsequent addition of polyvinyl polymers (Sta-Put or Nalco-Trol (Nalco Chemical Co.)) increased averaged drop size considerably, even at low polymer concentration. Increasing polymer concentration did not result in a further increase of drop sizes. The dynamic viscosity of the mixture increased with polymer concentration (up to 3.6 mPa·s for 0.04% Sta-Put; 6 mPa·s for 0.04% Nalco-Trol), but surface tension was not affected (about 29 $mN \cdot m^{-1}$ for all mixtures).

Zhu et al. (1997) investigated the effect of recirculating spray liquid in a test stand, with respect to polymer composition and drop size spectrum. Polymer adjuvants based on the common components polyacrylamide, polethylene oxide and a polysaccharide (xanthan gum) were used in different concentrations. All solutions showed degradation (decrease of volume median diameter) due to recirculation. It is reasonable to assume that degradation is due to breaking of the polymer chains under shear or extensional stress. It appeared that the longer the polymer chains were, the easier they were broken and the more rapidly degradation occurred. Solutions of non-ionic polymers degraded more rapidly than those of anionic polymers. For non-ionic polymers, regardless of molecular weight and concentration, volume median diameter decreased to about the value of pure water after circulating only two times. Solutions of anionic polymers appeared to be more resistant to degradation, especially at higher concentration and higher anionicity. This is probably due to the formation of a network of polymers, which is more resistant to shear stresses and prevents breakage of polymer chains. The top-down ranking of the polymers tested with respect to resistance to breakdown: xanthan gum, anionic polyacrylamides, non-ionic polyacrylamides, polyethylene oxide.

Liquids can be divided into solutions and emulsions. Oil-in-water (o/w) emulsions often behave similar to solutions with respect to viscosity, density and surface tension (Schmidt 1980). With the use of ‘invert emulsions’ (water-in-oil, w/o) much higher viscosities can be obtained (Hartley and Graham-Bryce, 1980)

Butler Ellis et al. (1997) investigated physical properties of aqueous sprays containing 0.5% Ethokem (cationic surfactant with polyoxyethylene tallow amine) or 0.5% LI-700 (acidifying surfactant with soyal phospholipids). The former adjuvant appeared to reduce drop size, the latter increased drop size. Using Phase-Doppler Anemometry (PDA) they found that the effect on drop size was related to the location inside the spray cloud: the changes were largest near the center of the cloud. Further, with Ethokem the average velocity of drops of a certain size appeared to be lower than with water, while with LI-700 drop velocities appeared to be higher than with water. Drops (>300 μm diameter) of the mixture containing Ethokem, when caught in oil trays, seemed to show air inclusions. Occasionally drops as small as 150 μm diameter also showed air inclusions. The authors assumed that the observed changes in drop size spectrum and liquid sheet geometry could be explained by dynamic changes in surface tension and surface viscosity in the ageing liquid sheet. More recently, Butler Ellis and Bradley (2002) found in wind tunnel experiments that the drop size reduction obtained with a surfactant of 0.5% Ethokem did not always lead to increased drift, but this seemed to depend also on nozzle type and wind speed. However, their experiments involved flat fan nozzles as well as hollow cone nozzles, which might have obscured a clear interpretation.

Spanoghe et al. (2001) measured the effect of two types of glyphosate formulations (RoundUp and Roundup Ultra) and the additive ammonium sulphate (Spanoghe et al., 2002) on drop size (Malvern) of a standard flat fan (XR110015VP). No effect was observed on spray quality.

For the adjuvants Tween 20, Agral 90, Silwet L77 and Break Thru the effect on spray quality (Volume Mediane Diameter; VMD) was measured for the nozzle types Teejet 8001vk, 80015vk, 8002vk, XR8003vk, XR8004vk, XR8005vk, XR8006vk, XR8008vk at a pressure of 2 and 3 bar. Tween 20 added to tap water resulted in no differences in VMD for the nozzle types 8001 - 8003. For the coarser nozzle types a finer spray was found when Tween 20 was added to the water compared to water alone. Silwet L77 added to water resulted in a coarser spray for all nozzles types. This effect was more pronounced at a concentration of 100 mg/l than at 1000 mg/l. Agral 90 coarsened the VMD for all nozzle types at a concentration of 100 mg/l. At a concentration of 1000 mg/l, Agral 90 had no effect on VMD for the nozzle types 8001-8003 and made VMD finer for the nozzle types 8004-8008. The same pattern was seen for Break Thru. At a concentration of 100 mg/l, Break Thru coarsened VMD for all nozzle types measured. At a concentration of 1000 mg/l, no effect was observed for the nozzle types 8001-8002 and VMD became finer for the nozzle types 8003-8008.

Butler Ellis and Tuck (2000) investigated the effect on drop size (PMS) of eight additives with three venturi type nozzles, a twin-fluid nozzle and a standard flat-fan nozzle. The additives used were mineral oil (Actipron), vegetal oil (Codacide oil), Polyethoxylated tallow amine surfactant (Ethokem), Polyethoxylated nonylphenol surfactant, Polyethoxylated heptamethyl trisiloxane (Silwet L-77), synthetic latex (Bond), Poly-1—p-menthene (pinolene; Clinger) and a modified soya licithine (Li-700).

The spray mixture had a significant effect on the spray quality. For nozzles of the same type, even nozzle design can lead to large differences. Additives react differently with venturi-type nozzles than with standard flat fan nozzles. For water-soluble additives, VMD increased for venturi type nozzles and decreased for the standard flat fan nozzle. For emulsions and dispersions, VMD increased for both venturi and flat fan nozzle types. Largest differences occurred for the twin-fluid nozzle. There was an increase in VMD of 20% with the Ethokem solution and a decrease in VMD of 8% for the Li-700 solution.

The effect of mixtures of three types of EC formulations (clodinafop-propargyl, trifluralin and cypermethrin), an SC formulation (isoproturon), a non-ionic surfactant, a methylated vegetal oil and a emulsifiable vegetal oil on drop size (Oxford Visisizer) of three types of venturi nozzles was investigated by Powell et al. (2002). All three nozzles produced a finer spray quality for the water + non-ionic surfactant compared to tap water alone. Differences in VMD are mostly larger between nozzle types than between mixtures for the same nozzle.

De Ruiter et al. (2003) report the effect of a flat fan, a pre-orifice flat fan and a venturi nozzle type on spray quality of a fluazinam (Shirlan) solution in combination with three additives. The spray quality was not changed because of the fluazinam. The additives gave different effects on drop size depending on nozzle type. Van de Zande et al. (2001) showed that even for fine sprays such as the Low Volume Mister used in greenhouses, the VMD of tap water (30 um) decreased by 20% (25um) when using a solution of the fungicide Fungaflor and the insecticide Decis. Holterman et al. (1998) found similar effects with the use of additives and tap water with standard flat fan nozzles (XR11004), an anvil flat fan nozzle (TT11004) and a venturi nozzle type (TD110-03). Calculated drift with the IDEFICS model (Holterman et al., 1997) showed large effects of additives coinciding with the effects of additives on the volume fraction of drops smaller than 100 μm.

Hewitt et al. (2000) mentioned the Dropkick model with which the effect of formulations and additives on drop-size can be calculated. The model is developed for aerial applications, but also includes some nozzles used in field applications. Hewitt et al. (2001) mentioned that dispersions generally have no effect on drop size. Emulsions result in coarser spray qualities because of an earlier break-up of the liquid sheet close to the nozzle outlet, also resulting in a narrower spectrum. For an 8002 nozzle type, VMD could change by as much as 20%. Also the proportion of small and large drops in the spray (SPAN) changed because of the different solutions.

Hewitt (2001) concluded that because of the different solutions, a nozzle could be classified one class smaller or coarser in the BCPC spray quality classification system (Southcombe et al., 1997) than based on classifications with tap water as spray solution. The effect of solution is different for the different nozzle types. More information is needed to advise on the effect of spray solution on spray quality.

Often no clear relationship can be given between changes in drop size and concentration of the agrochemical or adjuvant. For example, while drop size may increase at low concentrations, at higher concentrations the increased drop size may decrease to its initial value (Dexter, 2001). The type of spray nozzle used may be an essential factor as well, yet still not clarifying the observed effects (Spanoghe et al, 2002).

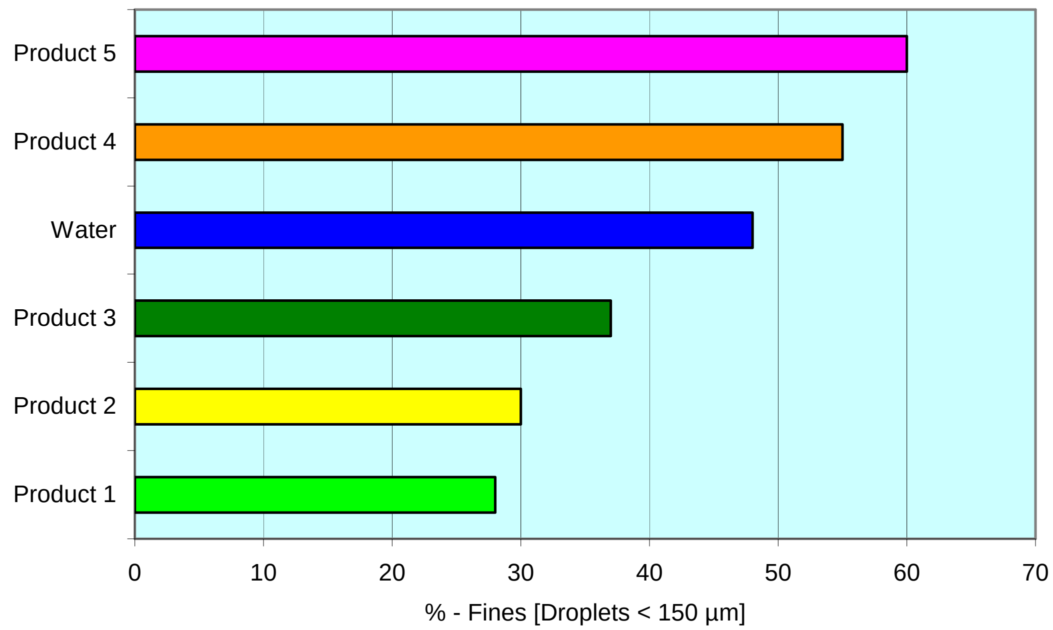
Formulations with drift reducing potential

Herbst (2003) measured drift potential in a wind tunnel and droplet size spectra for 11 types of spray nozzles and nine different spray liquids. A standard measurement protocol was used to characterise the driftability of the sprays by the Drift Potential Index (DIX; Herbst and Helck, 1998). A large influence of the spray liquid was found on the drift potential, although no general trend could be found even for spray liquids of the same formulation type. Changes in drift reduction class for a nozzle type could occur because of the change in formulation. A specific formulation could have opposing effects on driftability for a standard flat fan nozzle or a drift-reducing venturi type nozzle. These effects also differ depending on the concentration used. It is difficult to define a test liquid that is representative of real spray liquids. The DIX values for water are generally in the middle range of values for the formulations used in this study.

The physical properties of spray liquids could significantly influence horizontal and vertical drift profiles. Comparative tests have been performed in wind tunnels (Butler Elis and Bradley 2002, Stadler 2004) and field experiments. Spray droplet measurement with different formulations demonstrated that the number of fines could be significantly different for spraying with identical nozzles and spray conditions. Usually the mean volumetric diameter (MVD) represents the fineness of nozzles. More interesting for drift purposes is the content of fines that is produced while spraying. The content of fines has been checked with different formulations (containing more or less surfactants) at recommended concentration.

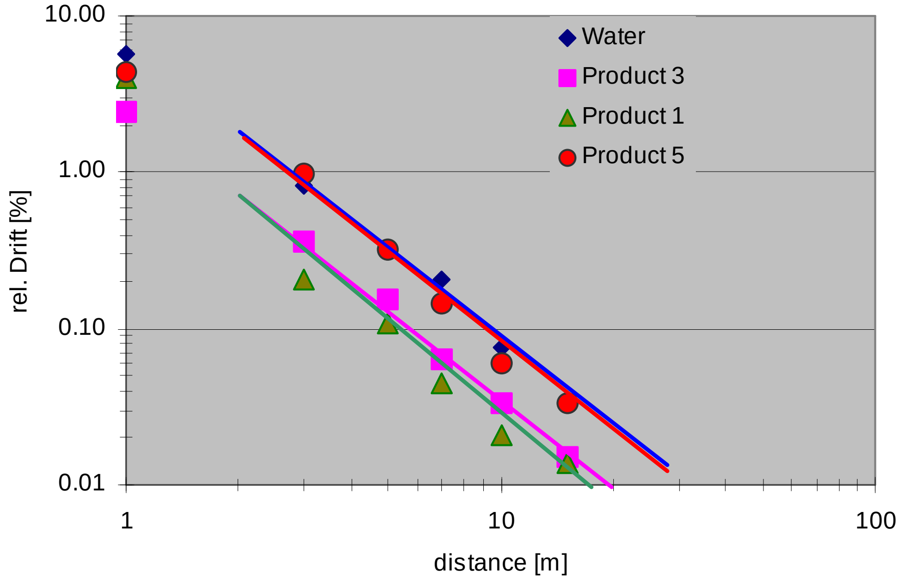
A laser scan device had been used to measure the droplet size (Figure 1.8). It is interesting that the content of fines can be enlarged in comparison to water and on the other hand can be significantly reduced. For one nozzle type and identical conditions, the number of fines can vary by more than 100%. This has been checked in wind tunnel experiments as well.

Figure 1.8. Percentage of fine droplets for different formulations (Nozzle LU 120 03; 3 bar; Stadler, 2004).



The correlation between fine droplet content and spray drift effects is obvious, but it cannot be used for pre-calculation of spray drift. Drift data in a wind tunnel resulting from a sample of the products in Figure 1.8 are compared in Figure 1.9. It is obvious that different additives or products reduce drift by up to 75% (product 1 and 3 compared to product 5 and water). On the other hand it is interesting that product 5 did not increase drift in comparison to water even though it had more fines in comparison to water in droplet size measurement.

Figure 1.9. Deposition on soil of different products (wind 3 m/s; nozzle LU12003; 2.5 bar; Stadler, 2004)



Stadler (2004) showed in both field experiments and wind tunnel measurements that an optimized formulation resulted in a 50% lower spray drift deposit compared to tap water. The differences in drift potential of the formulations were predominantly steered by the volume fraction of small drops in the spray. The higher the volume fraction of small drops (<150 µm), the higher the spray drift.

1.3.8.3 Conclusions

Formulations and tank additives affect spray quality. The effect of spray tank solution on drop size is different for the different nozzle types. Formulations and tank additives that make spray quality finer increase spray drift. Coarsening spray quality reduces spray drift. More information is needed to advise on the effect of the spray solution on spray quality and therefore spray drift.

1.3.9 Definition and measurement of spray drift reduction

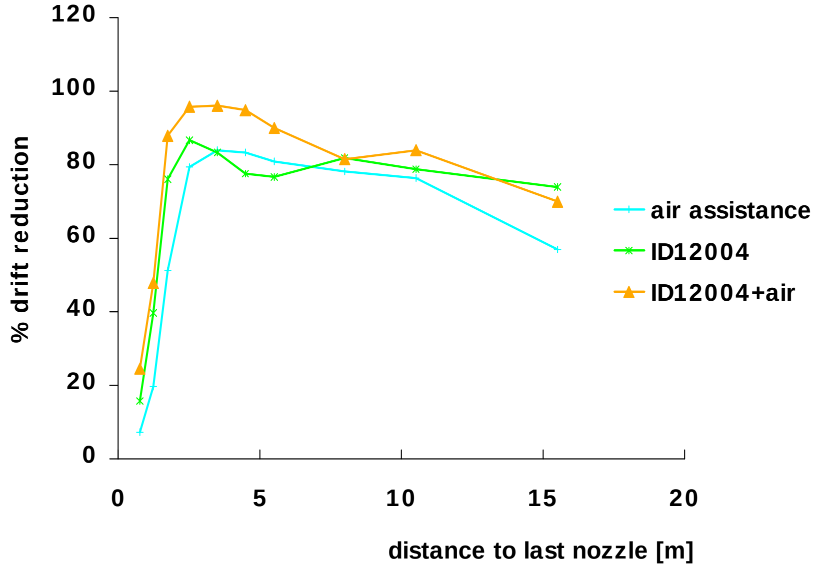
The effect of all kinds of parameters on spray drift is often expressed as spray drift reduction. The basis of this methodology is the definition of a reference situation against which to compare drift reducing measures. A well-defined reference situation is essential when discussing drift reduction.

1.3.9.1 Arable crops

In arable crop spraying, at least in Germany, the United Kingdom and the Netherlands, there seems to be a fairly similar reference sprayer. It is agreed that sprayer boom height should be set to 0.5m above crop canopy and the nozzle type is specified (Southcombe et al., 1997). Discussions on this subject have led to an effort to standardize spray drift measuring and evaluation (ISO, 2001; /TC23/SC6/WG4andWG7). In order to evaluate the effect on spray drift reduction, it is essential to compare the effect on spray drift deposition at identical distances. For example, spray drift reduction is evaluated in the Netherlands at a specified distance from the last nozzle (or the last tree row) coinciding with the water surface (2-3m) in the ditch (Huijsmans et al., 1997). In the UK, spray drift reduction is evaluated for a zone of 2-6m from the end of the sprayer boom (Gilbert, 2000). In Germany, Herbst and Ganzelmeier (2000) described the classification of sprayers in spray drift reduction classes based on an evaluation of the distance of 5-50m from the end of the sprayer boom. As spray drift reduction can vary with distance from the field edge (Figure 1.10), classification of a sprayer may differ from country to country although based on the same dataset.

The ISO standard on drift reduction classification suggests to evaluate drift classes on zones of 1-5, 5-10, 10-15, steps in between, 45-50 or 1-50m from the field edge. Drift reduction classes mentioned are 25, 50, 75, 90, 95 and 99% compared to a reference situation.

Figure 1.10. Spray drift reduction of air assistance and nozzle type compared to a standard flatfan nozzle (XR11004 @3bar) spraying a potato crop with a spray volume of 300 l/ha



1.3.9.2 Orchards

There is no agreement on a reference for an orchard sprayer, mainly because of large variations in orchard lay-out (row spacing, tree shape and sizes) between regions and therefore in adapted spray techniques.

1.3.10 Internationally- implemented drift mitigation measures

It seems that across Europe some consensus already exists about the classification of drift reduction. This is also found in almost agreed international standards developed within ISO on spray drift measurements (ISO/CD12057) and spray drift classification (ISO/CD22369). In the different countries, different entries are used to come to a spray drift reduction categorisation. An inventory of listed criteria is presented in Table 1.3 and Table 1.4.

Table 1.3. Entries for drift reduction in field crops in different countries

Drift reduction	Technique	Germany	UK	Netherlands	Sweden
50%	Nozzle-pressure-material	X	X	X	
	Twin-fluid nozzles	X	X	X	
	Spray quality				X
	Air assistance	X		X	X
	Boom height	X		X	X
	Sprayer speed	X		-	X
	Application zone width	X		X	X
	End-nozzle		X	X	
	Tunnel sprayer (bed-crops)			X	
	Släpduk			X	X
	Windbreak crop			X	
	Wind speed				X
	Air temperature				X
75%	Nozzle-pressure-material	X	X	X	
	Twin-fluid nozzles	X	X	X	
	Air assistance	X	X	X	
	Boom height + nozzle-type	X		X	
	Sprayer speed	X			
	Application zone width	X			
	End-nozzle			X	
	Shrouded boom		X		
	Släpduk			X	
	Band sprayer			X	
90%	Nozzle-pressure-material	X		X	
	Twin-fluid nozzles			X	
	Air assistance + nozzle-type			X	
	Boom height + nozzle-type + air assistance	X		X	
	Sprayer speed	X			
	Application zone width	X			
	Släpduk + nozzle-type			X	
	Vertical nozzle pipes (asparagus)	X			
	Band sprayer	X		X	
95%	Air assistance + nozzle-type			X	
	Släpduk + nozzle-type			X	
	Boom height + nozzle-type			X	
99%	Shielded bed sprayer			X	
	Släpduk + nozzle-type			X	

Table 1.4. Entries for drift reduction in orchards in different countries

Drift reduction	Technique	Germany	UK	Netherlands	Sweden
50%	Nozzle-pressure-material	X			
	Leaf-sensor	X		X	
	Shut-off air outside direction 5 rows	X			
	Maximal air capacity 5 rows	X			
	Maximal spray pressure 5 rows	X			
	Shut-off air outside direction 3 rows	X			
	Maximal air capacity 3 rows	X			
	Maximal spray pressure 3 rows	X			
	Application zone width	X			
	Hail net over entire orchard	X			
	Windbreak net on edge field			X	
	Shut-off spray outside direction last row			X	
75%	Nozzle-pressure-material	X			
	Leaf-sensor	X			
	Sprayer type (cross-flow fan)	X			
	Shut-off air outside direction 5 rows	X			
	Shut-off air outside direction 3 rows	X			
	Maximal air capacity 3 rows	X			
	Maximal spray pressure 3 rows	X			
	Application zone width	X			
	Max. fan capacity	X			
	Tunnel sprayer			X	
	Windbreak crop			X	
	Hail net over entire orchard	X			
90%	Nozzle-pressure-material	X			
	Sprayer type (cross-flow fan)	X			
	Shut-off air outside direction 5 rows	X			
	Maximal air capacity 5 rows	X			
	Maximal spray pressure 5 rows	X			
	Application zone width	X			
	Max. fan capacity	X			
	Max. crop height 2.2m	X			
	Max. row width 2.2m	X			
	Tunnel sprayer	X			
	Collector-recycling sprayer	X			
	Fan Air direction	X			
	Windbreak crop			X	
99%	Tunnel sprayer + nozzle type	X			

1.4 Risk mitigation for surface runoff and erosion

1.4.1 Introduction

1.4.1.1 Origin of runoff

Surface runoff can be divided into two distinct types according to the mechanism of initiation:

Hortonian runoff occurs when rainfall intensity exceeds the permeability of the soil surface. Its occurrence is influenced by numerous factors, with the most important being rain intensity, soil stability (sealing process) and soil cover (canopy effect).

Saturation runoff takes place when a poorly pervious layer is present in the near subsurface. The dominant climatic factor is the volume of rainfall rather than its intensity.

In the latter situation, topsoil saturation generates hypodermic runoff (lateral flow in the subsurface), which evacuates a part of the excess water (or all of it in the absence of surface runoff). This process is controlled by slope and soil porosity. Tile and ditch drainage accelerate hypodermic runoff and lower the temporary water table generated by the saturation. The FOCUS SWS did not distinguish the two types of runoff (they were taken together as runoff).

1.4.1.2 Runoff and erosion

Water has a shear stress effect on soil which is influenced by the slope and the thickness of the runoff flow across the soil surface. As a consequence, sheet and rill erosion may occur on sloping plane fields and the concentration of the flow in the talwegs generates ephemeral or permanent gullies, even in situations with a gentle slope, when the catchment area of the talweg is relatively non pervious and induces high runoff volumes. This situation is common on the silty plateaus in north-west Europe. It should be recognized when dealing with exposure of water bodies by plant protection products that several measures are taken by farmers to reduce erosion and intake of nutrients into water bodies.

1.4.1.3 Pesticide transfer in runoff

Pesticide transfer is a complex process arising from various interactions between the properties of the compound, climatic conditions, and the agricultural and environmental characteristics of each individual situation. Moreover, research programs on this topic have only been conducted over the last 15 years in Europe, so that the technical basis for corrective actions relies on incomplete scientific knowledge. The main question is not really the identification of the mechanisms involved, which are quite well known, but their quantification and their relative predominance.

However, some points of practical interest are well established:

- Sorption and persistence are the determining properties for pesticide transfer (Baker et al., 1994; Gril et al., 1999).
- Erosion may induce residue transfer of strongly sorbed molecules. Coarse soil particles have a much lower sorption capacity than the fine humus-rich ones so that pesticide concentrations are generally much higher in the thin surface soil layer than in deeper horizons. Sheet erosion which is difficult to observe may thus be more effective for pesticide transfer than spectacular gully erosion.
- Pesticide transfer is greatest when runoff occurs in the first weeks after the application because of degradation in soil and progressive fixation of chemicals within the soil matrix (Baker et al. 1995). For this reason, herbicides are a particular concern as their application often occurs on bare soil or with limited soil coverage in periods when runoff is the most likely (at least under European conditions).
- The rapidity of water movement on and through the soil to the water body is a factor which significantly increases the intensity of pesticide transfer.

1.4.1.4 Mitigation measures to control pesticide transfer in runoff

In principle the following issues could be considered when deciding upon risk mitigation measures to reduce the intake via runoff:

- Buffer strips
- Conservation tillage
- Incorporation
- Cover crops
- Contour planting
- Terracing

- Crop rotation, mixture crops
- Field size – patch spraying
- Minimizing soil surface compaction (good tillage practices)
- Settlement ponds (biobeds)
- Irrigation technique (drips vs furrow vs flood) – managing antecedent soil moisture conditions
- Reduced soil contamination from drift reducing techniques

Some correspond to tillage and cropping practices, the others are landscape management techniques. It is intended to summarise here the currently available information concerning the direct and indirect effects of the various measures on pesticide transfer via surface runoff. A general difficulty is to collect information usable for mitigation measures on well established and sufficiently validated scientific results. The discussion of possible mitigation measures below focuses primarily on techniques which are specific to transfer in runoff, assuming that implementation of general good practice is a prerequisite.

1.4.2 Vegetated buffer zones

The efficiency of vegetated buffer zones to trap sediment and fertilizers is well known and documented (for example, Dillaha et al. 1999). The main factor is the high infiltration rate of grass surfaces, in relation to the sealing protection effect of a dense cover, the good structure of the upper layer, and the development of roots. However, the experimental results concerning pesticide trapping of grassed buffer zones are more varied.

1.4.2.1 Assessment of pesticide trapping efficiency

At the moment, only grassed buffer zones have been significantly studied. However, the studies are recent, except the similar experiments of Asmussen et al. (1977) and Rhode et al. (1980) on a grassed waterway with 2,4-D and trifluralin, respectively. More information is available concerning sediment and nutrient removal.

Some reviews are available on the efficiency of grassed buffers in removing pesticide from runoff (Patty 1997, USDA 2000). The experiments are conducted under natural rainfall conditions (for example, Arora et al. 1996, Patty et al. 1997) or use simulated rainfall and/or runoff (Misra et al. 1994, Klöppel et al. 1997, Souiller et al. *in press*). A very recent review (Lacas, submitted) makes a point on the current knowledge concerning pesticide transfer in grassed buffers and was drawn upon in writing this section.

Main factors influencing pesticide transfer in vegetated buffer zones

It is difficult to summarise this information since measured pesticide removals are quite variable. A first question is to verify if there is a significant relationship between trapping efficiency and pesticide mobility. Table 1.5 collects the data available in 2005, presenting pesticide trapping efficiency in relation to strength of pesticided sorption (Koc) and buffer width. The variability of the trapping, even for one molecule and one publication, shows clearly that the adsorption property of a substance is not sufficient to predict pesticide transfer through a grassed buffer zone. In fact, the Koc (or other properties like solubility) is quite rarely identified as the main factor in interpretation of experimental studies (Table 1.6). The adsorption property of the molecule certainly plays a role, but the influence may be masked by other factors including infiltration and the time between application and runoff.

Table 1.5 Trapping efficiency of grassed buffer zones (adapted from USDA, 2000 and Patty, 1997)

Pesticide	Koc	Pesticide trapped (%)	Experimental support[1]	Buffer width (m)	Buffer area/source area (%)	Study reference
Permethrin	100000	27-83	RN+RF	7.5-15	9-18	Schmitt et al. 1999
Trifluralin	8000	86-96	RN	24		Rohde et al., 1980
Chlorpyrifos	6070	57-79	RN			Boyd et al., 1999
Chlorpyrifos	6070	62-99	RN	2.5-5		Cole et al., 1997
Pendimethalin	5000	77-100	RN	1-15		Spatz et al. 1997
Fenpropimorph	2770	42-100	RN	1-15		Spatz et al. 1997
Fenpropimorph	2770	71	RF	5		Syversen 2003
Diflufenican	1990	97	NC	6-18	12-48	Patty et al., 1997
Diflufenicanil	1990	82-98	RF	3		Souiller et al. 2002
Lindane	1100	72-100	NC	6-18	12-48	Patty et al., 1997
Propiconazole	949	63	RF	5		Syversen 2003
Glyphosate	750	39	RF	5		Syversen 2003
Norflurazon	600	65	NC	2-4	9-18	Rankins et al., 1998
Diuron	479	70-98	NC	3-6		L'Helgoualch 2000
Terbutylazine	306	29-100	RN	1-15		Spatz et al. 1997
Terbutylazine	306	35-65	RF	10		Kloeppel et al. 1997
Metolachlor	200	16-100	NC	20	3.3-6.7	Arora et al., 1996
Metolachlor	200	30-47	RN		3.3-6.7	Misra et al., 1996
Metolachlor	200	55-74	RN	2-4	9-18	Webster et al., 1996
Metolachlor	200	91-98	NC+RN			Tingle et al., 1998
Cyanazine	190	17-100	NC	20	3.3-6.7	Arora et al., 1996
Alachlor	170	>90 (grass 70)	NC	8 (grass) +41 (wood)		Lowrance et al., 1997
Dichlorprop	170	49-78	RF	10		Kloeppel et al. 1997
Acetochlor	150	56-67				Boyd et al., 1999

Pesticide	Koc	Pesticide trapped (%)	Experimental support[1]	Buffer width (m)	Buffer area/source area (%)	Study reference
Alachlor	122	10-61	RN+RF	7.5-15	9-18	Schmitt et al. 1999
Isproturon	120	99	NC	6-18	12-48	Patty et al., 1997
Isoproturon	120	18-90	RN	1-15		Spatz et al. 1997
Isoproturon	120	78-88	RF	10		Kloeppel et al. 1997
Isoproturon	120	62	RF	3		Souiller et al. 2002
Atrazine	100	8-100	NC	20	3.3-6.7	Arora et al., 1996
Atrazine	100	52-69				Boyd et al., 1999
Atrazine	100	91		6	27	Hall et al., 1983
Atrazine	100	30-57		9		Hoffman , 1995
Atrazine	100	97	NC	8 (grass) +41 (wood)		Lowrance et al., 1997
Atrazine	100	35-60		5-10	10-20	Mickelson et al., 1993
Atrazine	100	26-50	RN		3.3-6.7	Misra et al., 1996
Atrazine	100	44-100	NC	6-18	12-48	Patty et al., 1997
Atrazine	100	5-43	RN+RF	7.5-15	9-18	Schmitt et al. 1999
Atrazine	100	63-96	RF	3		Souiller et al. 2002
Fluormeturon	100	60	NC	2-4	9-18	Rankins et al., 1998
Mecoprop	85	97-100	RN	1-15		Spatz et al. 1997
Metribuzin	60	50-76	RN	2-4	9-18	Webster et al. 1996
Metribuzin	60	91-98	NC+RN	0.5-4	2-18	Tingle et al. 1998
Thiodicarbe	57	32-96	NC	3-6		L'Helgoualch 2000
Pirimicarb	53	23-100	RN	1-15		Spatz et al. 1997
Fosethyl-Al	45	36-95	NC	3-6		L'Helgoualch 2000
2,4-D	20	70	RN	24		Assmussen et al. 1977
Dicamba	2	90-100	RN	2.5-5		Cole et al., 1997

[1] Natural conditions (NC), Rain simulation (RN), Runoff simulation (RF)

Table 1.6. Main factors found to influence pesticide transfer through vegetated buffers in different studies

Study	Main factors influencing pesticide transfer
Arora et al., 1996	Infiltration, time between application and runoff
Cole et al., 1997	Infiltration and soil moisture, dilution, formulation
Kloeppel et al., 1997	Infiltration (in relation with strip width and inlet flow), dilution
Lowrance et al., 1997	Season, time between application and runoff
Misra et al., 1996	Infiltration, runoff concentration
Rankins et al., 1998	time between application and runoff
Patty et al., 1997	Infiltration, time between application and runoff
Schmitt et al., 1999	Grass age, Koc
Souiller et al., 2002	Infiltration, Koc

Following the review of the FOCUS Landscape and Mitigation report by the EFSA PPR panel (EFSA, 2006), literature reported in the interval between 2005 and 2007 was collated (Table 1.7) and added into the database.

Table 1.7. Trapping efficiency of grassed buffer zones for literature gathered since 2005

Pesticide	Koc	Pesticide trapped (%)	Experimental support[1]	Buffer width (m)	Buffer area/source area (%)	Study reference
Metolachlor	200	81-100	NC	6-12	30-42	Klein (2004)
Terbuthylazine	306	83-100	NC	6-12	30-42	Klein (2004)
Pendimethalin	5000	93-100	NC	6-12	30-42	Klein (2004)
Glyphosate	21700	91.3	NC	2	4	SWAP-CPP (2006)
AMPA	8000	50.0	NC	2	4	SWAP-CPP (2006)
Dimethomorph	348	28.6	NC	2	4	SWAP-CPP (2006)
Dichlobenil	171	33.3	NC	2	4	SWAP-CPP (2006)
Dithiocarbamates	>1000	57.1	NC	2	4	SWAP-CPP (2006)
Unnamed herbicide	<250	97.7	RN	5	29	Jones (1993)
Diuron	1067	81.8-98.4	RF	6	0.9	Lacas (2006)

[1] Natural conditions (NC), Rain simulation (RN), Runoff simulation (RF)

The current literature data only apply to situations where: (i) surface runoff enters the buffer as sheet flow (rather than as channelled flow), and (ii) the soil in the buffer is not saturated and the infiltration capacity of the buffer is not reduced by soil surface sealing. Furthermore, the experimental conditions of literature runoff studies may not be directly comparable to those in the field as they tend to be undertaken on small plots and often include artificial rainfall at high intensity or even artificial runoff. A straight-forward analysis of the data reported in Tables 1.5 and 1.7 is difficult because of the different experimental conditions and the measured variation in buffer efficacy for buffer zones of different sizes. There are also some references where the efficacy of the buffer can only be approximated. Furthermore, there are a significant number of studies from the US and Australia where it is difficult to determine the extent to which experimental conditions are relevant to the European situation.

In order to estimate the efficacy of vegetated buffer zones, all non-European data were removed from the database, leaving 9 publications reporting efficacy values for at least one compound. Next, the resulting database was split according to whether the compound was primarily present in the aqueous or sediment phase of runoff. Efficacy values were reported into the respective sub-sets where the original study reported separately on aqueous- and sediment-phase runoff. Studies that only reported pesticide loads in total runoff were placed into either the aqueous-phase subset (Koc < 1000 ml/g) or the sediment-phase subset (Koc > 1000 ml/g). A single value was determined for each buffer system/pesticide application (i.e. multiple events monitored for the same buffer/application were combined; different years or different pesticides monitored on the same buffer were reported separately). The arithmetic mean was calculated for a given buffer/pesticide/year where there were replicates. The resulting database contained 76 datapoints (11 compounds) for sediment-bound transport, and 107 datapoints (12 compounds) for aqueous-phase transport. These data are summarised in Tables 1.8 and 1.9. The number of datapoints for individual buffer widths is often very small, but the range and mean of data are reported for information. Note, that data for almost all buffer widths are significantly skewed towards higher reduction efficiencies (i.e. there are a large number of high values and only a small number of low values; see Figure 1.11 for an example).

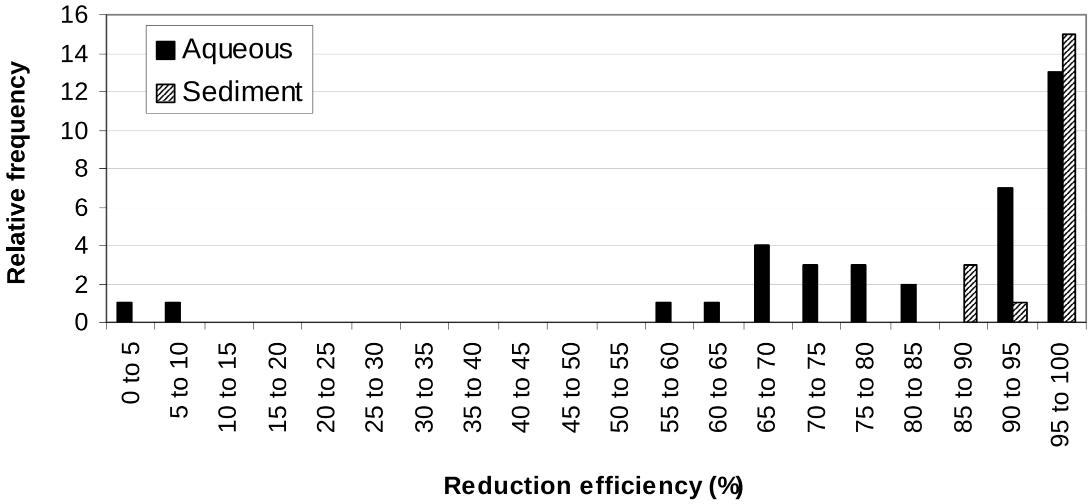
Table 1.8. Summary of European data for efficacy of pesticide removal from the aqueous-phase of runoff by vegetated buffer zones.

Buffer width (m)	n	minimum	maximum	mean
1	4	44.00	75.50	61.50
2	2	28.57	33.33	30.95
4	4	46.00	69.00	61.38
5	8	9.95	97.73	62.07
6	13	44.00	100.00	84.28
7	10	35.00	100.00	77.00
10	23	1.89	99.99	77.21
12	13	60.00	100.00	91.71
15	13	33.00	100.00	88.25
18	7	97.00	100.00	99.15
20	10	14.12	98.34	86.06
Total	107	1.89	100.00	80.22

Table 1.9. Summary of European data for efficacy of pesticide removal from the sediment-phase of runoff by vegetated buffer zones.

Buffer width (m)	n	minimum	maximum	mean
1	2	48.50	76.50	62.50
2	2	28.57	33.33	30.95
4	2	64.00	89.50	76.75
5	18	11.34	97.73	65.82
6	9	72.00	100.00	91.82
7	7	-27.00	100.00	64.53
10	10	85.62	99.17	95.12
12	9	94.00	100.00	98.87
15	6	43.00	100.00	88.88
18	3	99.90	100.00	99.97
20	8	93.21	100.00	97.16
Total	76	-27.00	100.00	82.30

Figure 1.11. Histogram for efficiencies of vegetated buffers of 10-12 m width in reducing pesticide loading in the aqueous (n = 36) and sediment (n = 19) phases of runoff



The data presented in Tables 1.8 and 1.9 show that the efficiency of a given width of vegetated buffer is greater in reducing mass of eroded sediment and associated pesticide than in reducing volume of runoff water and associated mass of pesticide in the aqueous phase. Generally, there is greater variability in the reduction efficiency for smaller buffer widths.

Although the data indicate that reduction efficiencies tend to increase with larger buffer widths, there are insufficient data points to derive an overall relationship. Thus analyses were undertaken on data for specific buffer widths with measurements combined into width intervals (e.g. 18-20 m) to provide a more robust estimate of descriptive statistics. Table 1.10 provides summary statistics for buffer widths of 10-12 m and 18-20 m. The 10th and 90th percentiles of the distribution were calculated assuming a Weibull distribution (cumulative relative frequency = rank/n+1) with linear interpolation between the two measured datapoints surrounding the required percentile. The 90th percentile worst-case value has often been incorporated into regulatory procedures for exposure assessment on the assumption that it provides a sufficient degree of conservatism.

Table 1.10. Summary information for efficiencies in reducing pesticide load for different widths of vegetated buffers and different phases of surface runoff

Phase of runoff	Pesticide in aqueous phase		Sediment-bound pesticide	
Buffer width (m)	10-12	18-20[a]	10-12	18-20
Statistics for reduction efficiency (all values in %)				
Range	1.9-100.0	14.1-100.0	85.6-100.0	93.2-100.0
Median	94.0	97.0	99.1	99.8
Mean	84.7	90.0	96.9	97.9
10th percentile (best case)	100.0	100.0	100.0	100.0
90th percentile (worst case)	60.7	80.6	86.8	93.5
n	36	30	19	11

[a] Values for 15-m buffer were consistent with those for 18-20 m buffer and were included in the analysis to give more robust statistics

<u>Infiltration in the buffer zone</u>

Infiltration of runoff into the buffer is the principal cause of the ability to trap pesticide. “Adsorption or other processes that reduced concentrations were believed to be active at greater herbicide concentrations, but were not dominant, reducing herbicide losses from 0 to less than 10 %, compared with 25 to 48 % due to infiltration” (Misra 1996). The percentage of runoff which infiltrates depends both on flow generated uphill (and thus on rain characteristics and surface, slope, permeability and roughness of the field) and on permeability and dimensions of the buffer.

The permeability of the surface layer of a grassed zone is generally very high, more than the lower layers which limit the infiltration rate. If local climatic, pedological and topographic conditions induce the saturation of this surface layer in the wet season, the efficiency of the buffer will be greatly reduced during this period.

Time between application and runoff

It is widely noted in pesticide transfer studies that the closer to application time that runoff occurs, the higher are the quantities exported outside the plot. It is noted in many buffer experiments that small concentrations in the inlet flow generate a better efficiency of the buffer than larger ones. This is only true for more or less equivalent flow rates. As an example, if the first flow after application infiltrates more or less totally (which is often the case), the efficiency may be 100% even with large concentrations in runoff.

Other factors

One experiment (Schmitt et al., 1999) compared an old grassed buffer (25 years) to a young one (2 years), with a significant difference in efficiency: the cause is a better infiltration rate in the first one, but perhaps also better adsorption properties.

A wider buffer favours infiltration and also the dispersion of concentrated flow. Width is a broadly used sizing parameter for buffer design and regulation. However, for runoff control (c.f. control of spray drift), it should be related to runoff flow, then to uphill characteristics.

Dilution by rain falling on the buffer influences concentrations, but not the mass of pesticide.

Cole et al. (1997) compared a granular and a wettable powder formulation of chlorpyrifos and found larger loadings of the wettable powder in runoff.

1.4.2.2 Wooded buffer zones

Wooded buffer zones have been studied by the Tifton University, Georgia (Lowrance et al., 1997; Vellidis, 2000). In fact, a combined buffer (grass + wood) representing natural conditions was tested. The 3-year study showed a good extraction efficiency for this combination, with a better efficiency per meter length for the grass than for the forest. Runoff simulation experiments (Gril et al., 2003) have also shown a higher infiltration rate in wooded buffers than in grassed ones.

1.4.2.3 Fate of pesticides infiltrated in the buffer

Most studies identify the role of infiltration in buffer efficiency, and yet few results are available concerning the fate of infiltrated pesticides. It may not be different from what happens under a cultivated field: a slow movement through the micropores and more rapid transport through any macropores. Lowrance (1997) and Velledis (2000) monitored a shallow watertable at the Tifton study site. Atrazine and alachlor peaks appeared quickly after some runoff events, but lateral movement of the watertable was slow, and peaks seen in the water table did not generally result in peaks in the stream. Rapid movement has been observed by Souiller (2002) in grassed buffers and by Gril (2003) in wooded ones. This type of process is probably general, resulting from the high infiltration rate generally observed in the top layer of buffers.

Flow through the surface layer supplies subsurface runoff, if it is present (which is frequent in situations where runoff is a significant pathway for movement of water). Then, in the case of riparian buffers, this transfer may be a route of contamination of the stream which is not taken in account by all the surface experiments.

1.4.2.4 Discussion on trapping efficiency of buffer zones

<u>European climatic conditions</u>

A first and obvious conclusion is that a buffer zone may trap a major part of pesticides tranferred by runoff and that the actual amount will vary with site and climatic conditions. Nevertheless, two important nuances should be brought out. First, many of the studies have been performed with rain or runoff simulation techniques, which are not really representative of natural conditions as they comprise heavy rain and runoff and constant concentrations of substances in simulated runoff. Secondly, most of the references with natural conditions have a US origin. For a similar frequency and duration, US rainfall has a much higher intensity than in a large part of Europe and, then, leads probably to a too pessimistic view of the extraction efficiency of buffers, at least in the north and west of the EU. The rare data obtained under natural rain conditions in Europe come from the west of France (Patty et al., 1997) and from Provence (L'Helgoualch 2000) and show rather favourable trapping efficiencies (always over 50% and often over 90%).

Peaks in concentration

A point to consider is the probable lag of time in the transfer through a buffer, in comparison with a direct transfer to the stream. This sort of “chromatographic” effect should produce a mitigating effect on peak concentrations, even without mass reduction. Unfortunately, there are no experimental data concerning this point.

Seasonal considerations

Since a large part of the buffer efficiency is due to infiltration, the worst performance will be obtained by a saturated soil and the best by a soil dry enough to absorb all the runoff during a rain event. When conditions are dry enough and rain events do not induce a water transfer down to the water table or the stream, the infiltration and the subsurface runoff will not lead to a rapid contamination. These observations perhaps present an opportunity for the introduction of buffers into mitigation measures with specifications in relation to the time of application (spring, summer or early autumn treatments).

1.4.2.5 Operating rules for buffer implementation

These conclusions are still scientifically limited, however they permit to propose practical information concerning the design of vegetated buffer zones for reducing pesticide losses in runoff. Two technical releases have been published in France (CORPEN, 1997) and the US (USDA, 2000) which give more or less similar recommendations.

1.4.2.6 General remarks

Buffers are not relevant if runoff is not significant. However, as pesticide concentration in runoff is normally much higher than concentration in infiltration flow, it does not mean that runoff has to be the main water pathway (20% of rainfall lost in hortonian runoff corresponds to a high runoff rate).

Hydraulic by-passes (rills, gullies, ditches) through the buffer zone can totally invalidate its efficacy (Figure 1.12). In the same way, “concentrated flow is the nemesis of pesticide trapping by buffers” (USDA, 2000). Implementing a buffer downhill of a tile-drained field is not appropriate, except where runoff and drainage flow are both significant; the “boulbenes” (silty hydromorphic soils of SW France, subject to intense storms in spring) are an example of this situation.

1.4.2.7 Location of buffer zones

Location is probably the most important aspect concerning buffer implementation, particularly in relation to the above hydraulic considerations. Actually, it is just a matter of engineering logic, nearly independent of our understanding of pesticide transfer - buffers have to be implemented where runoff from treated fields is present. Therefore, to locate them along the stream (as for control of spray drift) is not the only possibility (Figure 1.13). The density of concentrated flows is likely to be higher along the stream than uphill. USDA (2000) suggests to implement buffers preferably along first- or second-order streams rather than along higher-order ones.

For gully erosion, it may be necessary to establish wider buffer zones where runoff leaves the field whereas on other parts of the field no buffer zones are needed even though they may be directly adjacent to water bodies. As buffer zones are a landscape feature, the efficacy of buffer zones should not only be determined at the point where the runoff moves into the surface water but rather for a larger stretch.

1.4.2.8 Sizing the buffer zones

The size of a buffer (i.e. the length along the slope) depends principally on uphill flow and buffer slope. Again, a modelling tool would be useful. The empirical advice of USDA and CORPEN (Figure 1.14) is in the same order of size: i.e. about 10 to 20 m for sheet or shallow concentrated flow. An important concentrated flow must be intercepted by longer buffers such as a grassed waterway or a meadow, to be implemented (or preserved) along or across the talweg.

Figure 1.12. By pass in a watershed

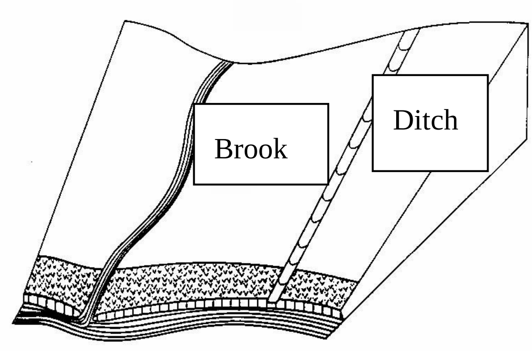


Figure 1.13. Location of grassed buffer zones

1 in the field 2 at the margin of the field

3 at the corner of the field 4 grassed waterway

5 meadow 6 along the riverside

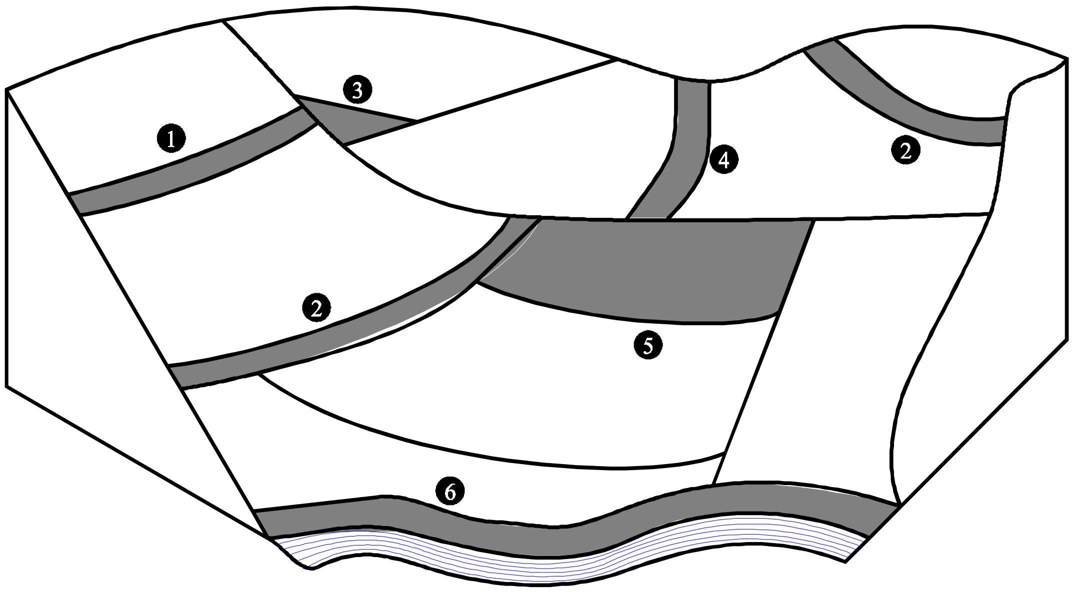
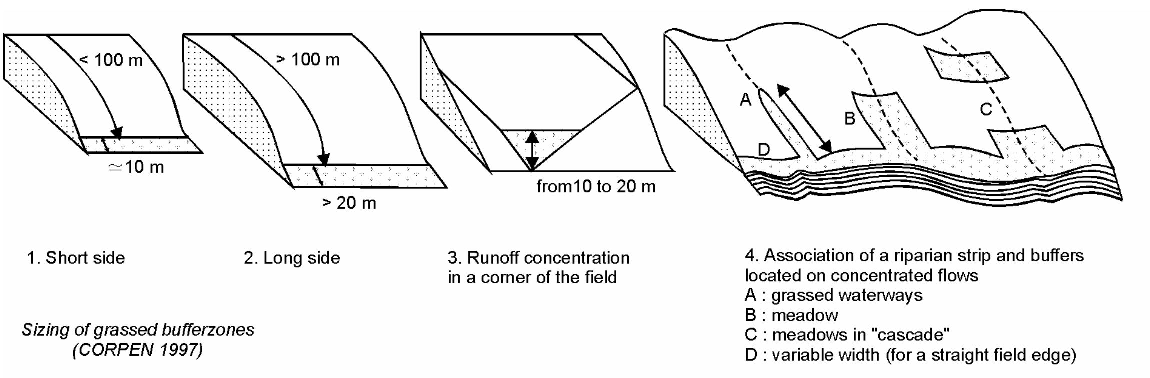


Figure 1.14. Different types and locations of buffer zones to reduce intake via runoff



1.4.3 Wetlands

Constructed wetlands or vegetated ditches have been proposed as risk mitigation techniques. Complementing their ecological importance as ecotones between land and water (Mitsch and Gosselink, 1993) and as habitats with great diversity and heterogeneity (Wetzel, 1993), specifically constructed wetlands are used extensively for water quality improvement. The concept of vegetation as a tool for contaminant mitigation (phytoremediation) is not new (Dietz and Schnoor, 2001). Many studies have evaluated the use of wetland plants to mitigate pollutants such as road runoff, metals, dairy wastes, and even municipal wastes (Brix, 1994; Cooper et al., 1995; Gray et al., 1990; Kadlec and Knight, 1996; Meulemann et al., 1990; Osterkamp et al., 1999; Scholes et al., 1998; Vymazal, 1990). According to Luckeydoo et al. (2002), the vital role of vegetation in processing water passing through wetlands is accomplished through biomass nutrient storage, sedimentation, and providing unique microhabitats for beneficial microbiological organisms. Macrophytes serve as filters by allowing contaminants to flow into plants and stems, which are then sorbed to macrophyte biofilms (Headley et al., 1998; Kadlec and Knight, 1996). Whether or not plants are capable of transferring contaminants from environmental matrices depends upon several factors including contaminant chemistry, plant tolerance to the contaminant, and sediment surrounding the plant (e.g. pH, redox, clay content; Zablotowicz and Hoagland, 1999).

Initially wetlands were employed mainly to treat point-source wastewater (Vymazal, 1990), followed later by an increased emphasis on nonpoint-source urban (Shutes et al., 1997) and agricultural runoff (Cole, 1998; Higgins et al., 1993; Rodgers Jr. et al., 1999). While the fate and retention of nutrients and sediments in wetlands are understood quite well, the same cannot be claimed for agrochemicals (Baker, 1993). Most of the available studies refer to the potential of wetlands for removal of herbicides and some other organic chemicals (Kadlec and Hey, 1994; Lewis et al., 1999; Moore et al., 2000; Wolverton and Harrison, 1975; Wolverton and McKown, 1976). Since wetlands have a pronounced ability to retain and process material, it seems reasonable that constructed wetlands, acting as buffer strips between agricultural areas and receiving surface waters, could mitigate the impact of pesticides in this runoff (Rodgers Jr. et al., 1999). The effectiveness of wetlands for reduction of hydrophobic chemicals (e.g. most insecticides) should be as high as for suspended particles and phosphorus, since these chemicals enter aquatic ecosystems mainly in particle-associated form following surface runoff (Ghadiri and Rose, 1991; Wauchope, 1978).

Table 1.11 summarizes the studies undertaken so far on insecticide retention in constructed wetlands and vegetated ditches. The initial studies attempting to quantify insecticide retention in wetlands by taking input and output measurements were carried out in South Africa with various insecticides. Schulz and Peall (2001) investigated the retention of azinphos-methyl, chlorpyrifos and endosulfan introduced during a single runoff event from fruit orchards into a 0.44-ha wetland. They found retention rates between 77% and 99% in terms of aqueous concentrations and >90% in terms of aqueous load. Particle-associated insecticide load was retained at almost 100% for all the studied organophosphate insecticides and endosulfan. A reduction in toxicity was also demonstrated for *Chironomus* sp. exposed *in situ* at the inlet and at the outlet (Table 1.11). Another study performed in the same wetland assessed contamination via spray drift of the most commonly used insecticide, azinphos-methyl, and found similar retention rates, although the retention rate for the pesticide load was only 54.1% (Schulz et al., 2001). In parallel, Moore et al. (2001) conducted research on the fate of lambda-cyhalothrin experimentally introduced into slow-flowing vegetated ditches in MS, USA. They reported a more than 99% reduction of pyrethroid levels below target water quality levels within a 50-m stretch due to an 87% sorption to plants. A further study investigated the fate and toxicity of chlorpyrifos using wetland mesocosms in Oxford, MS as well as the wetland in South Africa as a field example (Moore et al., 2002).

Another experiment in the Oxford mesocosms targeted the effects of vegetated versus non-vegetated wetlands on the transport and toxicity of parathion-methyl introduced to simulate a worst-case storm event (Schulz et al., 2002a). Both wetland invertebrate communities and *C. tentans* exposed in situ were used to illustrate positive effects from the presence of macrophytes (Table 1.11). The processes relevant for aqueous-phase dissipation of azinphos-methyl were the subject of another recent study using the flow-through wetland along one of the tributaries of the Lourens River in South Africa (Schulz et al., 2002b). The plants were shown to play an important role in the uptake of the chemical, but effects on the zooplankton communities were nevertheless detectable.

Apart from these more focused studies a few further studies are included in Table 1.11. The implementation of retention ponds in agricultural watersheds was examined by Scott et al. (1999) as one strategy to reduce the amount and toxicity of runoff-related insecticide pollution discharging into estuaries. However, wetland sizes and retention rates are not detailed further. Briggs et al. (1998) inferred a reduction of >99.9% in terms of the applied amount from a study in which nursery runoff was experimentally added to clay/gravel or grass beds of up to 91 m length (loadings not further quantified). A positive effect of settling ponds, situated below watercress beds in the UK that were not further described, was documented using mortality and acetylcholinesterase inhibition in *G. pulex* exposed *in situ* as endpoints (Crane et al., 1995b). Retention rates are not given, as the concentrations of malathion used in the watercress beds were not measured in this study.

In summary, only very few studies have dealt so far with wetlands or vegetated ditches as risk mitigation tools for nonpoint-source insecticide pollution. However, the results obtained so far on chemical retention and toxicity reductions are very promising (Table 1.11), and justify further investigations. A few other studies that have emphasized special aspects of pesticide fate or toxicity in wetlands (Dieter et al., 1996; Spongberg and Martin-Hayden, 1997) or uptake of insecticides to plants (Hand et al., 2001; Karen et al., 1998; Weinberger et al., 1982) corroborate the idea of wetlands for reduction of risk from insecticides.

Certain agricultural sectors, such as the greenhouse and nursery industry, have already started to adopt wetlands to treat pesticide-contaminated water (Berghage et al., 1999). In response to the historic wetland losses, the U.S. Department of Agriculture Natural Resource Conservation Service (USDA NRCS) has established four conservation practice standards (Codes 656, 657, 658, and 659) relating to constructed wetlands (USDA-NRCS, 2002). By establishing these practice standards, farmers and other agricultural landowners are given instructions on how to develop and use constructed wetlands as a best management practice to minimize nonpoint-source pollution of water bodies.

Table 1.11. Field studies on the effectiveness of constructed wetlands or vegetated ditches in mitigating insecticide contamination in surface waters.

Source	Substance	Inlet concentration	Retention Concn	Retention Load	Location	Wetland size	Dominant plant Species	Ecotoxicological assessment	Reference
Application to watercress beds	Malathion	-	-	-	Settling ponds below treated watercress beds	-	-	Mortality reduction, *Gammarus pulex* in situ bioassay	Crane et al. (1995b)
Experimental nursery runoff	Chlorpyrifos	No data	No data	>99.9 %†	Clay/gravel or grass beds below nursery, SC, USA	2 x 91 m	Cynodon dactylon	No data	Briggs et al. (1998)
Experimental runoff	Lambda-cyhalothrin	500 µg/L	>99%	>99%	Vegetated ditches, MS, USA	50 x 1.5 m	Polygonum amphibium, Leersia oryzoides, Sporobolus	No data	Moore et al. (2001)
Experimental runoff	Chlorpyrifos	73-733 µg/L	No data	83-98%	Wetland mesocosms, MS, USA	66 x 10 m	Juncus effusus, Leersia sp.	No data	Moore et al. (2002)
Experimental runoff	Methyl-Parathion	4-420 µg/L	>99%	>99%	Wetland mesocosms, MS, USA	50 x 5.5 m	Juncus effusus Leersia sp.	>90% toxicity reduction, *Chironomus* in situ bioassay, reduced effects on invertebrates	Schulz et al. (2002a)
Runoff	Azinphos-methyl Endosulfan Chlorpyrifos Azinphos-methyl Endosulfan Prothiofos	0.14-0.8 µg/L 0.07-0.2 µg/L 0.01-0.03 µg/L 1.2-43.3 µg/kg 0.2-31.4 µg/kg 0.8-6 µg/kg	77-93% >99% >99% >99% >99% >99%	>90% >90% >90% >99% >99% >99%	Flow-through wetland, Lourens River catchment, South Africa	134 x 36 m	Typha capensis, Juncus kraussii	>90% toxicity reduction *Chironomus* in situ bioassay	Schulz and Peall (2001)
Runoff	Azinphos-methyl Endosulfan Fenvalerate	0.2-3.9 µg/L 0.03-0.25 µg/L 0.05-0.9 µg/L	>99%‡ >60%‡ >80%‡	No data	Retention ponds, SC, USA	No data	No data	≈40% toxicity reduction, *Palaemonetes pugio* in situ bioassay	Scott et al. (1999)
Runoff	Chlorpyrifos	0.08-1.3 µg/L 2.6-89.4 µg/kg	>97% >99%	>97% >99%	Flow-through wetland, Lourens River, South Africa	134 x 36 m	Typha capensis, Juncus kraussii	>90% toxicity reduction *Chironomus* in situ bioassay	Moore et al. (2002)
Spraydrift	Azinphos-methyl	0.27-0.51 µg/L	90.1%	60.5%	Flow-through wetland, Lourens R., S. Africa	134 x 36 m	Typha capensis, Juncus kraussii	Reduced effects on zooplankton	Schulz et al. (2002b)
Spraydrift	Azinphos-methyl	0.36-0.87 µg/L	90.8%	54.1%	Flow-through wetland, Lourens River, South Africa	134 x 36 m	Typha capensis, Juncus kraussii	>90% toxicity reduction *Chironomus* in situ bioassay	Schulz et al. (2002b)

† Refers to the applied amount.

‡ Estimated retention since the concentrations refer to a catchment without ponds which was used for comparison.

1.4.4 Additional practices for runoff and erosion control

Theoretically, every technique aimed at increasing water infiltration or reducing soil losses has a beneficial effect in reducing pesticide concentrations in runoff water. Practically, a rapid review of the literature shows conflicting conclusions.

1.4.4.1 Conservation tillage

Most of the references originate from the US and deal with conservation tillage versus conventional practices (mouldboard plow or chisel). Conclusions may be difficult to extrapolate to European conditions for at least two reasons. First, the climatic conditions are quite different from European ones (at least in Northern Europe). As an example, a 50 mm 24h-rainfall has a return period of 2 years in Iowa and 50 years in Paris (Gril 1991). However, if tillage practices influence runoff, the effects will mainly be under conditions of 'ordinary' rain. Moreover, most of the US studies use rainfall simulation, which presents a worst case situation for the US and even more so for Europe. Secondly, the use of conservation tillage is much more developed in the US than in Europe. However, no-till or other similar techniques are expanding nowadays. Since the modification of the surface layer caused by non-conventional tillage is progressive, it is perhaps too early to draw definitive conclusions.

Additional considerations are:

1. Since most of the transfer occurs during the first runoff events after application, it is not the total annual effect of practices which is important, but what happens during the periods when application coincides with runoff likelihood: basically, after seedbed preparation in autumn or in spring. Thus, the potential effect of alternative tillage practices in Europe should mostly be studied in these periods.

2. Cover crops may influence runoff generation, although their effective action does not occur in the most strategic periods. However, the perennial crops (vines, orchards) are in a different situation; grass-sodding between rows has a very significant effect in limiting runoff and erosion (Gril et al. 1989). Moreover it reduced total loading of herbicides in runoff at the site. In fact, such a technique can be considered as a particular example of grassed buffer zones.

1.4.4.2 Modification of the application period

Changing the timing of treatment so that application periods do not coincide with periods when runoff is likely may reduce pesticide transport in runoff. Unfortunately, available experimental data are scarce. Simulated rainfall experiments in the US (Pantone et al. 1992) have shown a significant difference between transport of pre- and post emergence application of atrazine on maize.

On the experimental site of the technical institute for cereal and forage (ITCF) in La Jaillière (west of France) the monitoring of runoff and drainage transfer over about ten years shows differences regarding isoproturon and diflufenican transfer ratios between autumn and winter application (respectively before and during the drainage period). However, the interpretation of these results is still underway and conclusions have to be confirmed. Practically, herbicides are mostly likely to be the target for practices involving application timing.

<u>Soil incorporation and formulation type</u>

Pesticide incorporation in soil may contribute to make the compound less available for runoff, provided this practice is compatible with product efficacy. Wauchope (1978) indicated that transfer by runoff is greater for a wettable powder. Formulations can have a significant influence on run-off potential (Burgoa and Wauchope, 1995; Wauchope and Leonard, 1980; Leonard, 1990; Wauchope, 1978; Hartley and Graham-Bryce, 1980). Among the most significant formulation types with respect to impact potential are those formulations designed to limit the rate of release into soils (e.g. 'slow release' formulations). Such formulations may have the effect of extending the product's efficacy but, in doing so, may also extend its apparent persistence and availability for run-off. Certain surface applied formulations may be designed to increase availability (and, therefore, efficacy) at the soil surface but, in so doing, increase the potential availability for run-off. On the other hand, certain formulations are designed to reduce environmental impact.

1.4.5 Landscape management techniques

Theoretically, all non treated zones of a watershed in position to receive runoff water before it reaches the water body, may contribute in limiting pesticide transfer in runoff: buffer zones (grassed and wooded zones, wetlands), hedges and embankments, ditches. This contribution occurs by restricting water flow (by infiltration), settlement of sediment and sorption of residues. Practically, this mitigation effect may be either significant or not, depending on the infiltration capacity, the rapidity of the water transfer, the filtering capacity of the soil and the vegetation and their capacity to retain residues.

1.4.5.1 Conservation landscape management

Terraces and contour planting are designed (and very efficient) to control erosion. They also show a lesser level of efficacy in controlling runoff (Gril, 1991). Anyway, these techniques are generally difficult to apply in Europe because the shape of the fields is not adapted, for historical reasons except in specific cultures such as sloping vineyards.

Water and sediment control basins ("wascobs") are generally designed in association with terraces, to control gully formation, which has to do more with erosion problems than with pesticide transfer. Nevertheless, this sort of technique may be attractive in relation to buffer zones to convert concentrated flow into sheet flow (CORPEN 1997).

1.4.5.2 Crop patchwork and field size

A patchwork distribution of crops in the watershed, notably winter and summer crops, will limit runoff, erosion and pesticide transfer. Downhill fields act as buffers for the excess water from uphill fields. The effect is optimal if the downhill field is in a high stage of vegetation, and will be much less for bare soil conditions. However, it is statically better than a large area with crops in the same stage.

The effect of field size is unclear. Very large fields (i.e. 10 - 20 ha) have the drawback of large areas cultivated with a single crop, whereas very small fields show higher margin/surface ratios, with enhanced border effects.

1.4.6 Using measures to mitigate exposure via runoff in a regulatory context

A number of strategies are summarised in Table 1.12, based upon experience in managing run-off in the United States (SETAC, 1994). It is important to recognize that the strategies outlined here were proposed within an American regulatory context and may not be applicable within a European regulation framework. There are significant difficulties with enforceability with many of these techniques, but their adoption as recommended practices by farmers would undoubtedly reduce impacts significantly. Mitigation methods may have an impact on product efficacy and should be considered only with great care. Nonetheless, the information presented in Table 1.12 serves to illustrate the relative potential impact of different management options.

Table 1.12. Mitigation Practices Summary Guide* for Pesticide Run off Losses to Surface Water (SETAC, 1994)

Practice	Potential Reduction of Surface Run off Transport**		
	Strongly sorbed***	Weakly to moderately sorbed	Comments
Field Loss Reduction:			
Lower application rate	0-50%	0-50%	Loss reduction should be ≥ rate reduction; e.g., ¾ rate, loss should be reduced at 25%
Partial substitution	0-80%	0-80%	Environmental concerns may also exist for pesticide(s) used as substitute(s); upper range would go to 100% with total elimination of use
Partial treatment	0-75%	0-75%	*e.g.,* herbicide banding; loss or reduction in pest control and / or alternative treatments must be considered
Formulation	0-25%	0-50%	Potential effects need to be documented in field, laboratory, and / or modelling studies
Soil erodibiliy/special restriction	0-50%	0-25%	Restriction should be targeted to more strongly absorbed pesticides used on highly erodible land
Soil incorporation	25-50%	35-70%	Mechanical incorporation reduces the amount in surface mixing zone; more important for solution losses
Application timing	0-50%	0-50%	Loss decreases with time between application and storm run off; probabilistic weather information could be used
No-till	50-90%	0-40%	Erosion control by 90% feasible; run off reduction much less; herbicide wash off from residues may increase concentrations in run off
Conservation-tillage	40-75%	0-50%	Erosion control less than for no-till; run off reduction for first storm after application more reliable than for no-till
Surface drainage	0-20%	0-50%	Subsurface drainage can be reduced antecedent moisture and therefore run off and erosion; infiltration can reduce surface concentrations for less strongly absorbed pesticides
Avoid sealing./compacting	0-20%	0-50%	Very similar to the effects of infiltration differences caused by subsurface drainage
Irrigation	0-25%	0-50%	Improved management practices reduce run off and erosion; greater infiltration could reduce concentration for less strongly absorbed pesticides
Site cropping	0-75%	0-60%	Possible combination or reduced use (untreated strips) plus buffer effect (sediment deposition on contour)
Crop rotation	0-90%	0-90%	Pesticide needed could be much reduced in some rotations

* The rough estimates of the likely range of effects for each practice are based on limited research and/or professional judgement.

** It should be possible to predict a more narrow range for potential reduction using mathematical modelling for a specific set of soil and environmental conditions.

*** Partition coefficient typically >100

Table 1.12 (cont'd). Mitigation Practices Summary Guide* for Pesticide Run off Losses to Surface Water (SETAC, 1994)

Practice	Potential Reduction of Surface Run off Transport**		
	Strongly sorbed***	Weakly to moderately sorbed	Comments
Field-to-Stream Transport Reduction:			
Terrace/detention ponds	20-90%	5-20%	Sediment transport reduction; infiltration in basins could reduce volumes and therefore losses
Constructed wetlands	20-90%	0-50%	A practice for which little quantitative information exists
Buffer strips	10-40%	10-25%	Relative area untreated to total area important to be, $\leq$ 10%
Set-backs	0-50%	0-25%	Protection from spills (point-source) during mixing/loading/handling
Vegetative filter strip	20-60%	10-40%	To be effective, run off must pass through at nearly uniform depth; removal more efficient for lower contributing area-filter strip area ration
Grassed waterways	10-40%	2-10%	Similar to filter strip, but likely with higher contributing area-filter strip ratio; concentrated flow reduces effectiveness

* The rough estimates of the likely range of effects for each practice are based on limited research and/or professional judgement.

** It should be possible to predict a more narrow range for potential reduction using mathematical modelling for a specific set of soil and environmental conditions.

*** Partition coefficient typically >100

The mitigation efficiency of vegetative filter strips has been demonstrated in field studies. Such evidence has been used to assign benchmarks within exposure assessments conducted under national registration rules in Germany employing the EXPOSIT model (Winkler, 2001). Field studies that clearly indicate an influence of vegetative filter strips on the reduction of run-off were cited in the development of this registration tool. For example, comprehensive monitoring of terbuthylazine at four sites with high probability of heavy rainfall and with surface water bodies adjacent to corn fields - have shown that vegetative filter strips of 10 m width effectively protect surface water bodies from entry of terbuthylazine (no findings > 0.1 µg/L). Winkler also cites a study conducted by Klöppel et al. (1997) in which the retention of terbuthylazine, isoproturon and dichlorprop-P by grassed buffer strips of 10 and 20 m width was investigated. Even a 10 m wide grass strip gave a compound retention of approx. 90% (terbuthylazine 80 ± 11%, isoproturon 79 ± 12%, dichlorprop-P 74 ± 15%). For a buffer-strip of 20 m a maximum retention of 99% was observed (terbuthylazine 95 ± 4%, isoproturon 94 ± 5%, dichlorprop-P 92 ± 7%). In addition, there is a study by Real (1998), which shows that the introduction of grassed buffer strips is an effective method (run-off reduction by up to >99%) to prevent the entry of PPP in surface water bodies. Run-off of atrazine from a 1 ha sloped field was reduced by approx. 60% by a buffer strip of 6 m (approx. 70% for 12 m). A buffer strip of 18 m reduced run-off by approx. 97%. On this basis Winkler (2001) concluded that the framework for mitigation of run-off presented in Table 1.13 was justified:

Table 1.13. Summary of vegetative filter strip efficiency proposed by Winkler (2001).

Distance (m)	Vegetative filter strip (*Randstreifenbreite*) efficiency
0	0
5	50
10	90
20	97.5

In order to very crudely estimate reductions for any margin width, the following empirical relationship is considered valid over a limited margin width:

$$\text{Proportion Remaining (\%)} = 10^{(-0.083*\text{Margin width} + 2.00)}$$

EXPOSIT assumes that while a reduction in run-off volume reduces the mass of chemical entering the ditch, it also decreases the total volume of water (run-off water + resident ditch water) in which it is diluted in the destination water body. For this reason, a 50% reduction in run-off volume may not equate to a 50% reduction in PEC values. As a consequence, if this benchmark approach is employed as a basis for higher-tier modelling at Step 4, it is necessary to consider both a reduction in the mass of chemical and associated volume of water delivered into these destination water body. An example of an EXPOSIT calculation is provided in Table 1.14 to illustrate the impact of the volumetric adjustment:

Table 1.14. EXPOSIT Calculation to Illustrate Efficacy of Vegetative Filter Strips (VFS)

Vegetative filter strip width (m)	0 m	5 m	10 m	20 m
Vegetative filter strip efficiency (%)	0 %	50 %	90 %	97.5 %
Volume of run-off water (m^3)	100 m^3	50 m^3	10 m^3	2.5 m^3
Concentration in run-off water µg/l	3.64 µg/l	3.64 µg/l	3.64 µg/l	3.64 µg/l
Volume of ditch (m^3)	30 m^3	30 m^3	30 m^3	30 m^3
Combined volume (m^3)	130 m^3	80 m^3	40 m^3	32.5 m^3
Adjusted volume to address flowing water conditions (m^3) *	260 m^3	160	80 m^3	65 m^3
Final concentration (µg/l)	1.4 µg/l	1.14 µg/l	0.46 µg/l	0.14 µg/l

* A pragmatic adjustment factor of 2 is employed within EXPOSIT to address flowing water conditions

1.5 Risk mitigation for drainflow

1.5.1 Influence of soil type and pesticide properties

A literature review was undertaken to assess the influence of soil type and pesticide properties on leaching of pesticides to drains. The review encompassed studies on transport of pesticides to subsurface drains undertaken in Europe (it should be noted that the list may not be exhaustive and that many of the studies were from the UK). Experiments undertaken in the US were excluded. The minimum requirements for inclusion of a particular study were collection of samples of raw drainflow for analysis and the reporting of the maximum concentration and/or seasonal loss of pesticide in flow. Studies which assessed leaching through soil coring or where sampling focused on receiving surface waters were excluded. A unique record was assigned to each combination of field site, pesticide and calendar year. In total, 23 references were accessed (Table 1.15) giving 109 unique records for maximum concentration and 85 records for seasonal loss. Prior to analysis, the maximum observed concentration was standardised to the equivalent value assuming an application of 1000 g a.s. ha^{-1} (e.g. a concentration of 1 $\mu g\ l^{-1}$ for a pesticide applied at 100 g a.s. ha^{-1} was standardised to 10 $\mu g\ l^{-1}$).

The relationships between sand content of the various soils studied and either seasonal loss of pesticide to drains (Figure 1.15) or maximum pesticide concentration in drainflow (Figure 1.16) are plotted logarithmically to separate individual measurements. Both charts show large variability in measurements for different pesticides or different seasons at the same site (points aligned vertically). Monitoring has largely focused on sites with small sand content (i.e. large content of clay and/or silt). Nevertheless, there is a weak inverse relationship between sand content and both maximum concentration and seasonal loss of pesticide (Figure 1.15). Sites where sand content is small have large contents of silt and clay and it is assumed that these sites are most likely to have well-developed soil structure and significant potential for transport of pesticides via preferential flow. Seasonal losses of pesticides to drains range up to ca. 10% for sites with <10% sand, up to ca. 5% where the sand fraction is 10-20%, up to 2.5% for sand content 20-40% and less than 0.2% for sand content >40% (Figure 1.15). There is a similar pattern for maximum concentration in raw drainflow (Figure 1.16) with absolute maxima of up to 1000 $\mu g\ l^{-1}$ for sand content 0-20% and values decreasing by roughly an order of magnitude for each additional 20% sand.

Figure 1.15. Relationship between sand content and the seasonal loss of pesticide to drains (expressed as a percentage of that applied)

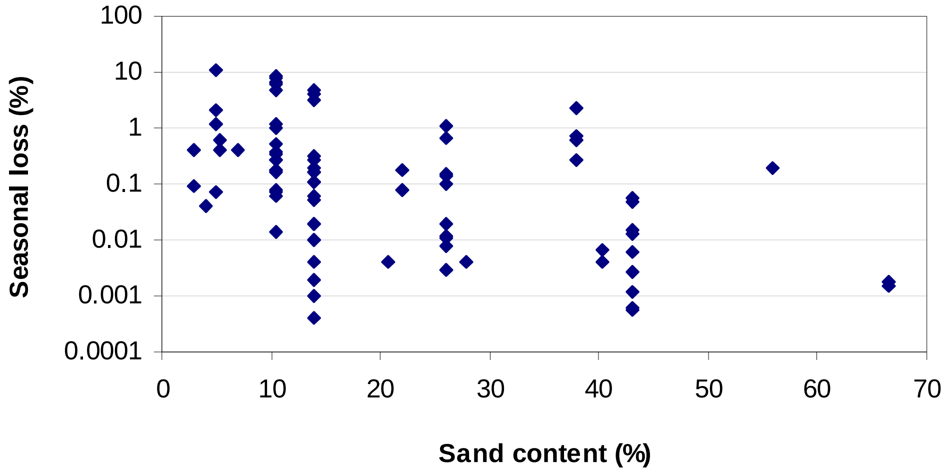
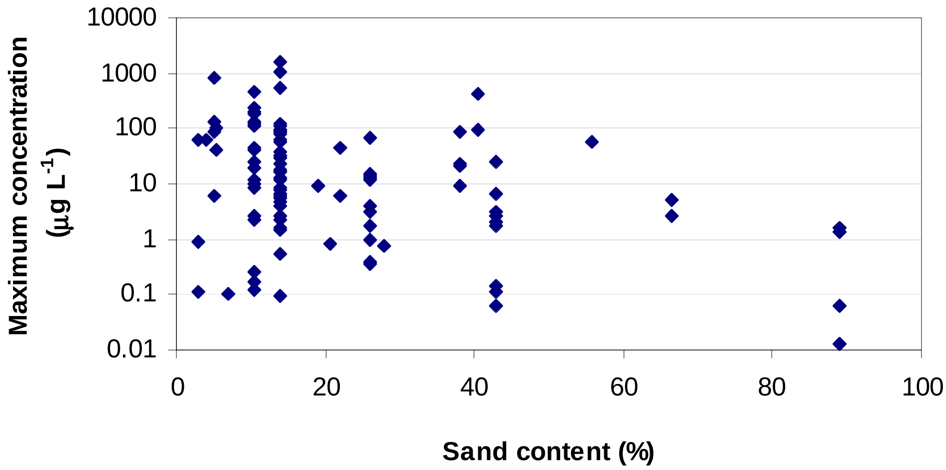


Figure 1.16. Relationship between sand content and the maximum concentration of pesticide measured in drainflow (standardised to an application rate of 1000 g a.s./ha



Next, the influence of pesticide sorption potential on losses to drains was evaluated. For each study, a literature value for Koc derived from the Agritox database (www.inra.fr/agritox) was combined with soil organic carbon content to calculate a site sorption coefficient (Kd). Figure 1.17 and Figure 1.18 show how the seasonal loss and maximum concentration of pesticides in drainflow varies with Kd for the range of European drainage studies. These charts ignore the influence on leaching of soil type and time between application and

Table 1.15. Summary of European studies on pesticide transport in drainflow (list may not be exhaustive)

Reference	Site	Field size	Texture	%OC	Years of study	Pesticides studied
Accinelli et al. (2002)	Po Valley, Italy	0.31	silty loam	0.85	1996-1997	Atrazine, metolachlor, prosulfuron, triasulfuron
Accinelli et al. (2002)	Po Valley, Italy	0.19	silty clay	0.74	1996-1997	Atrazine, metolachlor, prosulfuron, triasulfuron
Brown et al. (1995)	Cockle Park, Northumberland, UK	0.25	clay loam	2.70	1989-1991	Isoproturon, fonofos, mecoprop, trifluralin
Gatzweiler et al. (1999)	Südkirchen, Northrhine-Westfalia, Germany	1	sandy loam	1.30	1997-1998	Isoproturon
Gatzweiler et al. (1999)	Wöllstadt, Hesse, Germany	1	silty loamy sand	1.00	1997-1998	Isoproturon
Hardy (1997)	Stocklands NW, Boarded Barns, Ongar, UK	1.45	clay loam	1.50	1994-1995	Isoproturon, diflufenican
Hardy (1997)	Stocklands NE, Boarded Barns, Ongar, UK	1.6	clay loam	1.20	1994-1995	Isoproturon, diflufenican
Hardy (1997)	Fosters, Boarded Barns, Ongar, UK	1.3	clay	1.70	1994-1995	Isoproturon, diflufenican
Harris and Hollis (1998)	Knapwell Field, Boxworth, Cambs, UK	1.89	clay	2.20	1994-1997	Flutriafol, isoproturon, propiconazole, trifluralin
Harris and Hollis (1998)	Rosemaund, Herefordshire, UK	5.94	silty clay loam	1.70	1994-1997	Flutriafol, isoproturon, propiconazole, trifluralin
Harris and Pepper (1999)	Brimstone Farm, Oxon, UK	0.19	clay	3.60	1993-1999	Chlorotoluron, isoproturon, triasulfuron
Heppell et al. (1999)	Wytham, Oxon, UK		clay	2.57	1993-1995	Isoproturon
Johnson et al. (1994; 1995)	Wytham, Oxon, UK	0.18, 0.06	clay	2.57	1992-1993	Isoproturon
Kronvang et al. (2004)	Jutland, Denmark	0.28	clay	n.a.	2001	Bentazone, dimethoate, fenpropimorph, MCPA, pirimicarb, propiconazole
Novak et al. (2001)	La Bouzule, Lorraine, France	2.83	silt loam	1.36	1996-1998	Metolachlor
Novak et al. (2001)	La Bouzule, Lorraine, France	1.85	clay	1.90		Metolachlor
Peterson et al. (2002)	Near Copenhagen, Denmark	0.16	sandy loam	3.10	1999-2001	Ioxynil, pendimethalin
Smelt et al. (2003)	Central Netherlands	n.a.	silty clay loam	2.10	1998	Imidacloprid
Traub-Eberhard et al. (1995)	Soester Börde I, Nordrhein Westphalen, Germany	n.a.	silt loam	1.00	1992-1993	Chloridazon, isoproturon, metamitron, pendimethalin
Traub-Eberhard et al. (1995)	Soester Börde II, Nordrhein WP, Germany	n.a.	silt loam	1.20	1992-1993	Isoproturon, pendimethalin
Traub-Eberhard et al. (1995)	Brandenburg, nr Berlin, Germany	n.a.	sand	2.50	1992-1993	Isoproturon, metolachlor, pendimethalin, terbuthylazine
Villholth et al. (2000)	Gelbæk, Central Jutland, Denmark	0.0025	sandy loam	1.55	1997	Prochloraz
Williams et al. (1996)	Rosemaund, Herefordshire, UK	5.94	silty clay loam	1.70	1989-1993	Aldicarb, atrazine, carbofuran, chlorpyrifos, dimethoate, fenpropimorph, isoproturon, lindane, linuron, MCPA, trifluralin
Zehe and Flühler (2001)	Spechtacker, Weiherbach, SW Germany	0.09	silt loam	0.80	1997	Isoproturon

drainflow. As a consequence, the relationship between Kd and maximum concentration is rather weak. The negative relationship between Kd and seasonal loss to drains is stronger. Exceptional losses of up to 10% of applied pesticide are observed for Kd <1 ml g^{-1}, up to 5% for 1<Kd<10 ml g^{-1} and generally <0.2% for Kd >10 ml g^{-1}.

Figure 1.17. Relationship between pesticide Kd and the seasonal loss of pesticide to drains (expressed as a percentage of that applied)

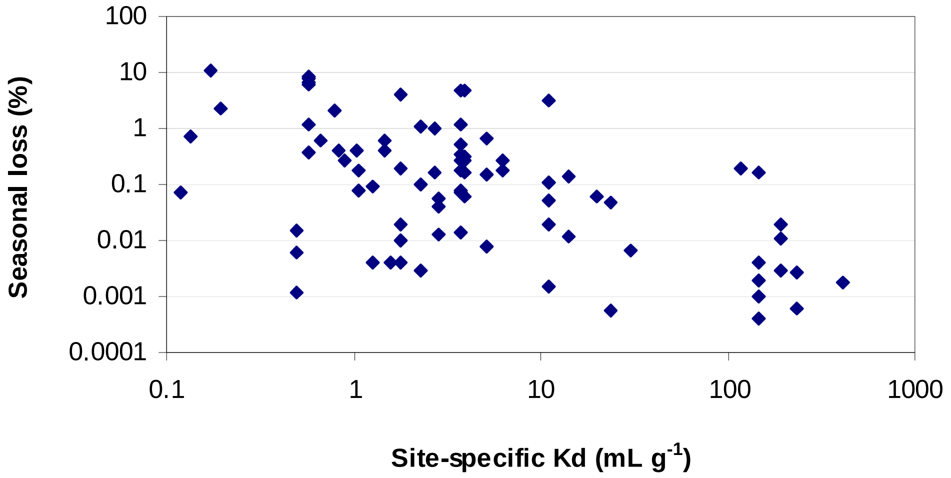


Figure 1.18. Relationship between pesticide Kd and the maximum concentration of pesticide measured in drainflow (standardised to an application rate of 1000 g a.s./ha

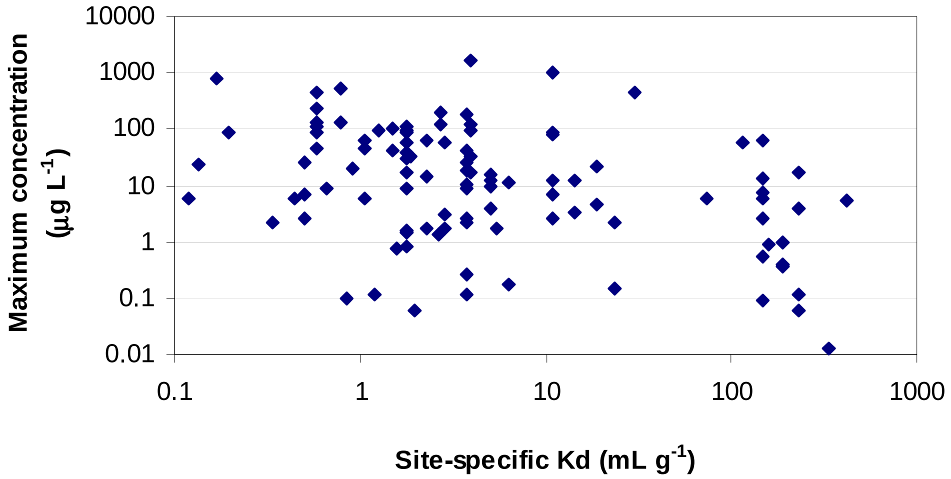


Figure 1.19 and Figure 1.20 present regression analyses which predict seasonal loss of pesticide to drains (Figure 1.19) or maximum concentration in drainflow (Figure 1.20) on the

basis of sand content of the soil and sorption coefficient for the pesticide (fraction of organic carbon in the topsoil multiplied by Koc). The datasets are not normally distributed, so natural logarithms were taken prior to the analysis. Both regressions are highly significant (P <0.001) although they account for only part of the variability in the data. The regression for seasonal loss of pesticide to drains predicts 39% of the variability even though no account is taken of factors such as season of application, interception by the crop, soil hydraulic parameters, type of drainage system, time lag between application and drainflow, wetness of the season or duration and intensity of rainfall events. The regression for maximum concentration is poorer (17% of the variability predicted), indicating that the factors excluded from the regression play a more important role in controlling peak concentrations in drainflow.

Figure 1.19. Multiple linear regression using soil sand content and pesticide Kd to predict the seasonal loss of pesticide to drains (expressed as a percentage of that applied); the line gives the 1:1 relationship.

$$\ln(\%LOSS) = 1.922 - 1.225.\ln(\%SAND) - 0.600.\ln(Kd)$$

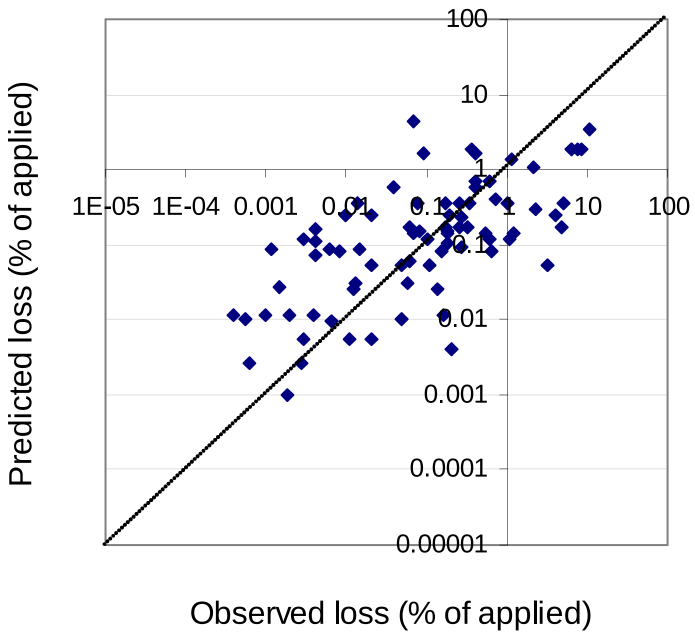
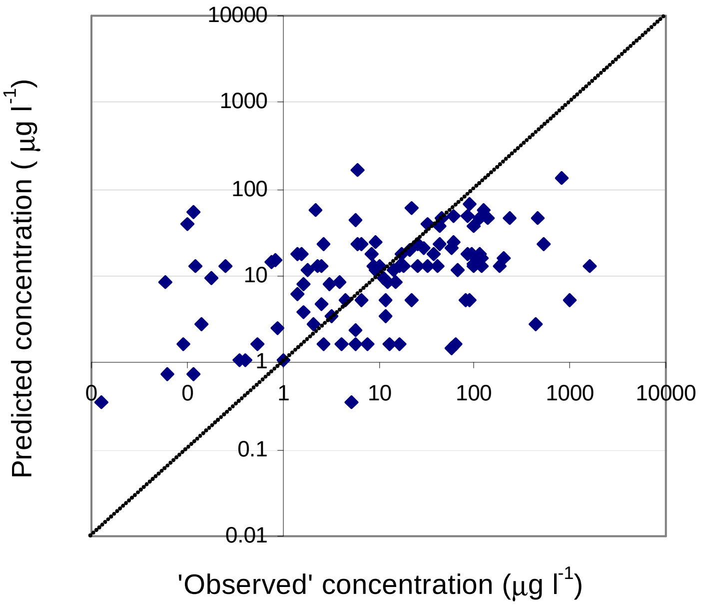


Figure 1.20. Multiple linear regressions using soil sand content and pesticide Kd to predict the maximum concentration of pesticide measured in drainflow (values normalised to 1000 g a.s. ha^{-1}); the line gives the 1:1 relationship.

$$\ln(MAXCONC) = 4.614 - 0.589.\ln(\%SAND) - 0.458.\ln(Kd)$$



1.5.1.1 Relationship to results from drainage studies in the US

Kladivko et al. (2001) provide a comprehensive review of more than 30 studies in the US on pesticide transport to subsurface tile drains. Many of the common features identified for the US experiments are also found in European studies. Transport to drains appears to be dominated by preferential flow. Peaks in concentration can be high but are usually short-lived. Highest concentrations are almost always found during the first drainage event(s) after application, with only a few studies demonstrating multiple concentration peaks over successive events. Where studies run over several years, there is large inter-season variability in peak concentration and total losses of pesticides, reflecting variability in patterns of weather.

The total losses of pesticide to drains in European studies span a larger range than those in the US. Kladivko et al. conclude that losses in US experiments are almost always less than 0.5% of the mass applied and frequently <0.1%. Only in exceptional studies are losses observed in the range 0.5-3%. In contrast, total mass losses in European studies are often in the range 0.1-1.0% of applied and results from five experiments show losses of between 1 and 10%. Part of the reason for the difference relates to the predominance of studies on clay soils in Europe whereas US sites are more evenly distributed between soil textures. Transport of pesticides via preferential flow will generally be most extensive in highly structured clay soils. Most pesticide applications during the US experiments were made in spring giving only a short period of drainflow before the onset of summer deficits. In contrast, treatment of European sites was most often made in autumn to winter crops and pesticide residues were exposed to a longer period over which transport via drainflow might occur. Eight of the applications in European studies resulting in losses greater than 1% of applied were made in autumn and four in spring. Eleven of the 12 applications were herbicides (atrazine, flutriafol, isoproturon, metolachlor, prosulfuron, triasulfuron) with the exception being propiconazole applied to the clay soil at Boxworth, UK.

1.5.2 Current regulatory status

Prior to the introduction of the FOCUS surface water scenarios, exposure of non-target aquatic organisms via drainflow has not been a routine component of ecological risk assessment to support Annex I inclusion. Drainflow was considered as a route of exposure within the national assessment procedures of some countries including Germany, Sweden and the UK.

There are only two risk mitigation options for drainflow which can be included within ecological risk assessment at the present time. In some countries, these options have already been applied (Germany, UK), whilst in others (e.g. Denmark) there is in principle recognition that they could be used (primarily by extrapolation from approaches to mitigate risk of leaching to groundwater).

1.5.2.1 Lower application rate

A lower application rate will result in a proportionately lower exposure via drainflow (e.g. Jones et al., 1995). The applicant would have to accept the lower application rate for the proposed use throughout the Member State as there are no mechanisms in place to allow differential rates according to whether or not a field is drained. The notifier would need to demonstrate that the lower rate still had sufficient efficacy.

1.5.2.2 Restriction in the application window

Losses of pesticides to drains are closely controlled by the time between application and the initiation of drainflow (Jones et al., 2000). Thus limiting applications to times when the drains are unlikely to be flowing (early autumn or spring) is an effective mitigation option even for moderately persistent compounds. Again, it would be necessary to demonstrate that efficacy would not be adversely affected.

In principle, mitigation options (1) and (2) could be combined. Thus a product could be approved with a lower application rate for a more vulnerable application timing and full rate for a less vulnerable timing.

In Germany, there is a special restriction for isoproturon and terbuthylazine where studies on transport via drainage systems have shown differences in losses for applications in autumn and spring. A label phrase relating only to drained land is applied if Annex VI TER triggers are not met: "Not to be used on drained surfaces between 1 June and 1 March."

Season-specific restrictions on the application window are the only type that have been implemented in the UK to date. A number of sulfonylurea herbicides have recently been approved for application in spring whereas autumn treatment is prohibited.

1.5.3 Potential alternatives to currently used risk mitigation measures

The sub-headings below describe further options that may be possible to mitigate risk of transport via drainflow. All of these currently lie outside of the regulatory system. At this stage, there is no consideration of the practicability or implications (e.g. for efficacy) of the various options.

1.5.3.1 Restriction of use to non-drained land

The simplest mitigation option would be to prohibit use of a compound on any field which has been artificially drained. For information, this option was discussed in the UK but was rejected on the basis that there was insufficient information available to farmers to allow a definitive decision on presence of drains (many drainage systems installed in the first half of the 20th century are still in place and functional, but maps are often missing). In principle the situation is the same in Germany. Even if it is known where a drainage system is present, it is difficult to predict how effectively it is working at the present time. A further constraint is that it is the inherent soil conditions (seasonal water logging within soil layers) that provide the potential for rapid transport of pesticides to surface waters. Whilst field drainage is frequently a prerequisite for arable cultivation in such soils, precluding application to drained land would not completely eliminate rapid transport to surface waters by subsurface lateral flow.

1.5.3.2 Restriction of use to low vulnerability land

Research suggests that pesticide losses via drainflow will be greatest on heavy clay soils with extensive macropores and significant potential for transport via preferential flow (Section 1.5.1). Certain soil properties may also constitute a greater vulnerability to losses via drainflow (e.g. alkaline soils for acidic compounds). Risk could be mitigated by prohibiting application on the most vulnerable soils.

1.5.3.3 Formulation

There is no specific information on effect of formulation on losses of pesticides via field drains. It can be anticipated that slow-release formulations which have been shown to reduce leaching of pesticides to groundwater (Flury, 1996) may also mitigate against transport via drainflow. Brown et al. (1995) reported a potential for transport of microencapsulated formulations through macropores in soil.

1.5.3.4 Soil management

A considerable amount of work in the UK has looked at various options to manage soil so as to reduce pesticide transport via drainflow. Brown et al. (2001) showed that generation of a fine topsoil tilth prior to application reduced losses by ca. 30% in drained lysimeters. The effects was explained on the basis of laboratory experiments which showed that the time for isoproturon to reach adsorption equilibrium in a clay soil increased with aggregate size (Walker et al., 1999). Novak et al. (2001) showed a similar response for transport of atrazine and trifluralin through soil columns repacked with aggregates of different sizes (1-5, 5-13, 13-20 and >20 mm diameter). Leaching was reduced in the finest aggregates of a clay soil by up to a factor of four, whereas there was little effect of aggregate size in a loamy soil. Data from a long-term field experiment with a heavy clay soil showed a possible relationship between topsoil tilth and leaching losses (again with ca. 30% reduction in leached load in soil with a finer tilth). However, heterogeneity between experimental plots was too great to demonstrate a significant effect (Jones et al., 1995; Harris and Catt, 1999).

Different cultivation practices are better developed in the US than in Europe and no-till practices are used widely. A significant number of studies have compared pesticide losses to drains under no-till and conventional (moldboard) ploughing and a smaller number have also compared the practice of ridge tillage. The soils investigated in these experiments include clays, clay loams, silty clay loams and loams. Results indicate that no-tillage practices either have no effect on pesticide losses to drains (Gaynor et al., 1992; Buhler et al., 1993; Logan et al., 1994; Baker et al., 1995 cited in Kladivko et al., 2001; Kanwar et al., 1999) or give increased losses relative to conventional ploughing (Kanwar et al., 1993; 1997 cited in Kladivko et al., 2001; Gaynor et al., 1995; Rothstein et al., 1996). Similarly, effects of ridge tillage on pesticide transport compared to conventional tillage have either been negligible (Baker et al., 1995 cited in Kladivko et al., 2001) or slightly deleterious (Kanwar et al., 1993; 1997; cited in Kladivko et al., 2001).

A soil sealant (Vinamul 3270 water based emulsion of a vinyl acetate copolymer) was applied to soil at Brimstone Farm (370 L/ha) following pesticide application. The aim was to decrease macropore flow by plugging soil cracks and decreasing infiltration of water into the soil. There were indications of a reduction in pesticide losses from sealed soil although results were not conclusive (Harris and Catt, 2000).

1.5.3.5 Soil incorporation

A trial at Brimstone Farm suggested that incorporation of pesticide into topsoil following application had no effect on subsequent losses in drainflow (Jones et al., 1995).

1.5.3.6 Application timing in relation to soil conditions

Field data demonstrate large losses of pesticides when application is made to very wet soil (because drainflow is likely soon after application) or to dry clay soils with extensive cracking (because transport via cracks under intense rainfall conditions can be very rapid). For example, Brown et al. (1995) showed larger leaching losses of isoproturon and mecoprop from dry soil after a spring application to a clay loam than from an autumn application to wet soil, presumably because of enhanced potential for macropore flow under drier conditions. These field observations are supported by laboratory experiments (Shipitalo et al., 1990; Edwards et al., 1993) and field studies with lighter soils (Isensee and Sadeghi, 1993; Flury et al., 1995) which showed that leaching losses were generally greater when irrigation or leaching rainfall was applied direct to dry soil rather than to soil which had been gently pre-wetted. However, antecedent soil moisture conditions *per se* were found to have no significant influence on losses of isoproturon via drainflow following autumn application to soil with differing moisture status (Brown et al., 2001). In the UK, an advisory label phrase has been applied to isoproturon: “Do not apply to dry, cracked or waterlogged soils as rain in these situations will move isoproturon too quickly below or across the surface, beyond the optimum site for weed control, and possibly into drains.”

Evidence of ageing has been demonstrated in laboratory experiments where more and more aggressive techniques are required to extract compounds as residence time of the compound in the soil increases (Hatzinger and Alexander, 1995; Burauel and Führ, 2000; Olesen et al., 2001). Pignatello et al. (1993) showed that mobility of freshly injected atrazine and metolachlor in repacked soil columns was greater than that of the same naturally aged compounds. There is evidence from field studies that suggests that ageing can significantly influence transport of pesticides in drainflow. Results from experiments at Brimstone Farm (Oxfordshire, UK) showed that for a cracking clay soil (Denchworth series), losses of isoproturon (a moderately mobile, moderately persistent herbicide) in drainflow decreased rapidly as time from application to the time when drainflow was initiated increased (Jones et al., 2000). This temporal decrease in isoproturon loads was faster than could be explained by loss mechanisms such as degradation and/or volatilisation. The effects of increased time between application and drainflow were less marked for more mobile and less persistent compounds such as triasulfuron.

1.5.3.7 Modifications to drainage design

Several studies have shown that concentrations of pesticides leaving treated fields in surface runoff are significantly larger than those in drainflow (e.g. Harris et al., 1994; Brown et al., 1995; Gaynor et al., 1995). Study on a clay loam soil in Northumberland showed that losses of four pesticides (mecoprop, isoproturon, fonofos, trifluralin) in surface runoff/interlayer flow from an undrained plot were always larger than those from an adjacent plot with mole drains (generally by a factor of 1.5 – 3) (Brown et al., 1995).

Although drainage reduces diffuse pollution by restricting surface runoff, it is generally acknowledged that much of the drained arable land in Europe is “over-drained” (i.e. drainage systems exceed the minimum required to provide effective control of the soil water regime). Pesticide losses to drains generally increase with efficiency of the drainage system and efficiency depends upon drain type, spacing and time since installation, as well as the properties of the soil itself. Thus Harris et al. (1994) concluded that drainage systems that minimise both surface runoff and rapid bypass flow to the drainage system would be the best compromise for water quality.

Work at Brimstone Farm looked at drainage restrictors (rotatable U-bends at the drain outlet) to raise the water table and delay the onset of drainflow. The strategy was effective in reducing losses of isoproturon (ca. 25% reduction), but had little effect on transport of the more mobile triasulfuron (Jones et al., 1995; Harris and Catt, 1999). Gaynor et al. (2002) compared effects of three water management systems on losses of atrazine, metribuzin and metolachlor in surface runoff and drainflow from a clay loam soil in Ontario. The treatments of free drainage (i.e. unrestricted), controlled drainage (similar to the restrictors imposed at Brimstone Farm) and controlled drainage with subsurface irrigation (not relevant to Europe) had no consistent effect on total herbicide losses in surface runoff and drainflow.

A study comparing effects of drain spacing (5, 10 and 20 m spacing) on diffuse pollution from a clay loam soil in Indiana showed that total losses of pesticides, nutrients, sediment and water were greatest for drains spaced 5 m apart and least for the 20-m spacing (Kladivko et al., 1991).

Removal of pesticides once they have entered drainflow would be a further mitigation option. Mole drains at Brimstone Farm were plugged with a highly sorptive, carbonaceous waste product (Harris and Catt, 1999). Results of laboratory and field studies showed that the plugs were highly effective in removing pesticides from draining water prior to entry into surface water. However, the sorption capacity of the material was finite and the binding was reversible, so a very large plug and/or replacement would be required to prevent elution of a significant pulse of pesticide once sorption capacity was exceeded.

1.5.3.8 Miscellaneous

Mackay et al. (2002) recently compiled a list of potential options to mitigate transport via drainflow in the UK. As well as some of the options set out above, these included partial substitution with a different compound, partial treatment (e.g. herbicide banding), modifications to site cropping (e.g. to incorporate untreated strips) and modifications to crop rotations (to reduce amounts of pesticide required). Evidence from studies in the US suggests that banding of herbicide treatment may be effective in reducing concentrations where practicable. Typical reductions in field application rates may be 50 to 66% and studies suggest that a similar or greater reduction in concentrations in drainflow can be achieved (Baker et al, 1995; Kanwar et al., 1999 cited in Kladivko et al., 2001).

1.6 Mitigating influences applying to all exposure routes

Running waters are differentiated from static[1] systems in the UK. Pesticide loads are quickly diluted in running waters. At the EPIF-workshop (http://homepage.mac.com/matthiasliess/EPiF/EPiFworkshop.htm), it was mentioned that in monitoring studies it is usually difficult to measures peak concentrations in lotic systems because dilution is fast. Only event-driven sampling strategies help to overcome this problem. Toxicity data are usually derived from tests lasting 48 hours at minimum in lentic systems. Work is required to develop harmonized approaches to differentiate between lentic and lotic systems when setting risk mitigation measures.

Chapter 3 of Volume 2 of this report summarises evidence that the external recovery potential of the area where the water body is located is very important when considering the occurrence of effects. This conclusion was also reached at the EPIF workshop. The more uncontaminated zones that are present in a water body or a catchment, the higher is the potential for recolonization by many organisms. Attempts should be made to consider this aspect when setting risk mitigation measures. Information from a suitable GIS can be used in order to make risk maps available to farmers.

Data from a monitoring study conducted in the north of Germany in an area with intensive agriculture and lots of small drainage ditches showed that there were no differences in the community of aquatic organisms between ditches located directly adjacent to fields with intensive pesticide use and those in areas with meadows where no products were used. The ditches were characterised by high stress for organisms due to considerable changes of the water level (often even falling dry), the temperatures and insolation (Sönnichsen, 2002). It may therefore be appropriate to differentiate between small man-made ditches where high “non-pesticidal” stress prevails and more natural water bodies when setting risk mitigation measures. Furthermore, the term ditch should be defined in an appropriate way. The ‘Dutch

[1] Lentic systems are static waters such as ponds, lakes and reservoirs; these are distinguished from lotic (flowing) waters such as streams and rivers.

ditch', which is used for example in the FOCUS surface water scenarios, is obviously not such a type of ditch.

The data to decide upon the risk reducing potential of local risk mitigating conditions (e.g. running water body) are not as extensive as for example those which form the basis for mitigation of spray-drift. However, in any case it is possible to decide on the base of conservative estimates. This type of restriction is more related to the realistic risk prevailing in a special use situation. On the other hand, these type of restrictions are more difficult to understand and more difficult to enforce. Training programmes and simple decision-making schemes for farmers are needed to keep them informed. If farmer organizations, industry and authorities responsible for controlling farmers support such approaches, it is clearly recommended to use such differentiated risk mitigation measures.

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2 RECOMMENDATIONS FOR STEP 4 AND REVIEWS OF STATE-OF-THE-ART IN EXPOSURE MODELLING

2.1 Modelling of aquatic exposure

2.1.1 Introduction and scale considerations

FOCUS Step 1 calculations incorporate a number of simplifying assumptions that are progressively refined in Steps 2 and 3 through the use of additional input data and model complexity as well as increased time and effort to perform the required calculations. The major assumptions and limitations incorporated in each step of the FOCUS surface water calculations are summarised in Table 2.1-1.

Table 2.1-1 Comparison of key assumptions in FOCUS Steps 1, 2 and 3

Factor	Step 1	Step 2	Step 3
Chemical application			
Application rates	Annual appln rate used	Individual appln events simulated	Individual appln events simulated (max = 8)
Application timing	No timing specified	3 seasons simulated	Actual appln dates simulated
Application intervals	No interval simulated	Intervals included in simulations	Intervals included in simulations
Application locations	No location specified	Northern and southern EU zones simulated	Up to 10 application sites simulated
Chemical data			
Solubility	Included	Included	Included as f (T)
Koc	Included	Included	Included, Freundlich
DT50 water/sediment	Included	Included	Included as f (T)
DT50 water	Not included	Included	Included as f (T)
DT50 sediment	Not included	Included	Included as f (T)
DT50 soil	Not included	Included	Included as f (T, Θ)
Wash off	Not included	Not included	Included as f (solubility)
Foliar dissipation	Not included	Not included	Included
Metabolites	Included (model as equivalent parent)	Included (model as equivalent parent)	Included (simulated in MACRO and PRZM, model as equivalent parent in TOXSWA)

Factor	Step 1	Step 2	Step 3
Crop			
Crop type	Included, 29 types	Included, 29 types	Included, 29 types
Canopy interception	Not included	Included, 4 values provided	Included, linked with crop growth model
Plant uptake of chem	Not included	Not included	Included
Agronomics			
Cropping dates	Not included	Not included	Included for each crop
Topographic factors	Not included	Not included	Included for each scenario
Climate data			
Precipitation	Not included	Not included	Daily values used
Temperature	Not included	Not included	Daily values used
Evapotranspiration	Not included	Not included	Daily values used
Irrigation	Not included	Not included	Irrigation scheduling used w/ 30mm events
Soil data			
Profile characterisation	Not included	Not included	10 scenario profiles
Organic carbon	Not included	Not included	10 scenario profiles
Moisture content	Not included	Not included	10 scenario profiles
Bulk density	Not included	Not included	10 scenario profiles
Drift			
Source of data	BBA	BBA	BBA
Buffer zone	1-3m, varying by crop	1-3m, varying by crop	1-6m, varying by crop
Drift loading	0-33.2%, f(crop)	0-33.2%, f (crop, applns)	0-23.6%, f(crop, applns)
Wind direction	directly from crop to water	directly from crop to water	directly from crop to water
Drainage			
Loading	10% instantaneous loading, combined with runoff	0-5% instantaneous loading, combined with runoff, f (region)	0-~3% time-distributed loading, f (scenario)
Timing	Same time as drift	4 d after last appln	Varies, determined by weather data
Delivery location	Edge-of-field into water	Edge-of-field into water	Edge-of-field into water
Buffer zone	Not included	Not included	Not included
Runoff / erosion			
Loading	10% instantaneous loading, combined with runoff	0-5% instantaneous loading, combined with runoff, f (region)	0-~3%, time-distributed loading, f(scenario)
Timing	Same time as drift	2 d after last appln	Varies, determined by weather data
Delivery location	Edge-of-field into water	Edge-of-field into water	Edge-of-field into water
Buffer zone	Not included	Not included	Not included

Factor	Step 1	Step 2	Step 3
Aquatic fate			
Water body types	Ditch	Ditch	Ditch, pond, stream
Hydrology	Static	Static	Dynamic
Depth of water body	30 cm	30 cm	Varies with water body and with time
Land:water ratio	10:1	10:1	100:1; 5:1; 100:1
Length of simulation	100 days	100 days	12-16 months
Scenario			
Field size	dimensions not fixed	dimensions not fixed	0.45-1.0 ha
Water body size	dimensions not fixed	dimensions not fixed	D, S = 1m x 100m, P = 30m x 30m
Field:wtr body ratio	10:1	10:1	D, S = 100:1, P = 5:1
Catchment size	no catchment assumed	no catchment assumed	D,S, P = 2, 100, 0.45 ha
% catchment treated	no catchment assumed	no catchment assumed	D,S, P = 0, 20, 100%
Extent of baseflow	not applicable	not applicable	minimal due to small size of catchment
Timing of catchment loading	not applicable	not applicable	simultaneous with edge-of-field loading
Effect of buffer zone	determines default drift values with no effect on runoff or drainage	determines default drift values with no effect on runoff or drainage	determines default drift values with no effect on runoff or drainage
Location of fields in upgradient catchment	not applicable	not applicable	immediately adjacent to water

Notes: T = temperature, Θ = soil moisture, D = ditch, S = stream, P = pond

The Step 3 calculations recommended by FOCUS consist of a series of single year calculations for up to ten individual modelling scenarios which represent potential aquatic exposures resulting from spray drift, erosion, runoff and/or drainage in a wide range of European settings. Each scenario consists of a single combination of chemical use pattern, soil profile, cropping agronomics and climatic data which is then combined with appropriate local surface water hydrology to provide PEC values for both surface water and sediment.

The goal of performing Step 4 surface water modelling is to add additional realism to the many simplifying assumptions that have been included in the Step 3 assumptions. A list of possible Step 4 refinements is provided in Table 2.1-2, grouped by the model that is primarily affected by the proposed change. As additional refinements are made to the Step 3 scenarios, it is important to consider how simulated results compare with any high quality measurements within relevant usage areas to ensure that the predicted PEC values fall within the range of observed concentrations and are sufficiently conservative for regulatory evaluations. Comparisons with monitoring data should be made with care as placing results within a clear temporal, spatial and usage context can be difficult.

Table 2.1-2 Key parameters in FOCUS Step 3 which are subject to refinement in Step 4

Climatic data: PRZM, MACRO, SWASH

Factor	Value assigned by FOCUS	Issue with FOCUS Value	Factor(s) to change	Source of Data to Support Change
Irrigation: timing and amounts added to met files for each crop	30mm irrigation events, scheduled on demand basis	Some current irrigation events can generate runoff, resulting in preventable runoff events	Change irrigation scheduling to more closely match demand or vary irrigation to minimise runoff	Published literature on irrigation scheduling; irrigation experts

Drift: EU Drift calculator, TOXSWA

Factor	Value assigned by FOCUS	Issue with FOCUS Value	Factor(s) to change	Source of Data to Support Factor Change
Drift values	BBA 2000 data are recommended by FOCUS	Other drift datasets are available in some countries	Substitution of alternative drift data	BBA, IMAG, Spray Drift Task Force, others
Distance between crop and field (buffer distance)	Conservative (i.e. short) buffer distances have been assumed for use in FOCUS Steps 1, 2 and 3	Actual buffer distances are typically much greater than the values set by FOCUS	Changing the width of the non-treated buffer zone between crop and adjacent surface water will reduce drift loading	The FOCUS Drift Calculator can be used to obtain drift loadings for alternative buffer distances.

Drainage: MACRO

Factor	Value assigned by FOCUS	Issue with FOCUS Value	Factor(s) to change	Source of Data to Support Factor Change
Chemical inputs	Chemical behaviour is currently characterised by single values of both various efate parameters as well as transport parameters (uptake and washoff)	Current FOCUS guidance is to use mean or median values for chemical inputs. A refined assessment could use refined chemical inputs or ranges of values with probabilistic modelling	Chemical inputs likely to have the greatest impact on leaching include: soil half-life sorption coeff, foliar washoff, foliar half-life	Experimental compound-specific data
Soil profile	Soil profiles with specific properties are recommended by FOCUS	Soil profiles in actual use areas may vary considerably from the idealised profiles selected by FOCUS for certain crops	Change soil profile(s) to match major profiles in actual use areas	Soil databases, literature, field measurements, local experts
MACRO default parameters	For orchards the factor RPIN is set to “deep”, which assumes that 60% of the total root density is found in the top 25% of total root depth (typically 20–25 cm)	Orchards (e.g. olives, citrus, pomefruit) are likely to have highest root density at layers deeper than 25 cm, potentially affecting leaching simulations	Change RPIN to a lower value for the top 25% of total root depth for orchard scenarios.	Literature, common agricultural practices (e.g harrowing of topsoil, which actively promotes the formation of deeper roots)
Selection of simulation year	A 50th percentile year was selected on the basis of simulations with winter wheat	The 50th percentile simulation year will vary as a function of the type of crop and annual precipitation/irrigation patterns	Test runs of alternative years can determine whether the selected simulation period reflects the 50th percentile water balance for a given crop	Test runs with MACRO to select a suitable 50th percentile year for a given crop.

Runoff and erosion: PRZM

Factor	Value assigned by FOCUS	Issue with FOCUS Value	Factor(s) to change	Source of Data to Support Factor Change
Chemical inputs	Chemical behaviour is currently characterised by single values of both various efate parameters as well as transport parameters (uptake and washoff)	Current FOCUS guidance is to use mean or median values for chemical inputs. A refined assessment could use refined chemical inputs or ranges of values with probabilistic modelling	Chemical inputs likely to have the greatest impact on leaching include: half-life in soil sorption coefficient for foliar applns: washoff coeff foliar half-life	Experimental compound-specific data
Selection of cropping dates	Generic cropping dates have been assigned by FOCUS	Actual cropping dates may vary from the dates assigned by FOCUS	Change cropping dates to reflect local practice	Agronomic experts, field development personnel, literature.
Selection of simulation year	A 50th percentile year was selected on the basis of runoff and erosion from two crops: winter wheat (non-irrigated) and maize (irrigated)	The 50th percentile runoff and erosion year will vary as a function of the type of crop and the annual precipitation pattern.	Test runs of alternative years can determine whether the selected simulation period reflects the 50th percentile water balance for a given crop	Test runs with PRZM to select a suitable 50th percentile year for a given crop.
Attenuation of runoff and erosion due to buffer effects	Edge-of-field runoff and erosion are assumed to be delivered directly to water bodies with no influence of the non-treated buffer zone	Non-treated buffer zones do not currently affect the runoff or erosion results in current FOCUS calculations	Attenuation of runoff and/or erosion may occur as a function of buffer width, depending upon the physical/chemical characteristics of a chemical. The P2T files produced by PRZM can be modified to reflect these reductions.	Literature, field experiments, agronomic experts. Also, see Section 1.4 on mitigation of runoff
Use of probabilistic modelling	Current FOCUS process uses single input values and provides a deterministic estimate of exposure	Stochastic evaluation will provide increased understanding of the influence of key input parameters, including weather	Chemical, agronomic, soil and climatic inputs	Literature, local experts
Formulations	PRZM does not directly consider the effects of formulation	Certain formulation-related properties may affect the potential for runoff and erosion	Change timing of application (e.g. for controlled release), type of application, etc.	Compound-specific data

1 Aquatic fate: TOXSWA

Factor	Value assigned by FOCUS	Issue with FOCUS Value	Factor(s) to change	Source of Data to Support Factor Change
Catchment size for FOCUS stream	Catchment size of 100 ha was assumed by FOCUS for the stream scenario	A 100 ha catchment is too small to support a permanent stream; as a result, the current scenario requires an outlet weir to maintain a water depth of 30 cm. In addition, as a simplification, the treated fields in the catchment are assumed to discharge into surface water at the same concentration and timing as the 1 ha treated field	The catchment size should be increased to a size sufficient to support a permanent 1m stream. This change is also likely to provide increased base flow and can incorporate the attenuating effect of upgradient fields being located at various distances from adjacent surface water.	Use maps and/or GIS to determine actual catchment sizes associated with first order streams, ditches and small ponds Use hydrologic calculations to determine catchment size necessary to sustain permanent streams via drainage, runoff and subsurface flow. Consult with hydrologic experts, FOCUS members and compare results with available surface water monitoring studies.
Characteristics of land area in catchments associated with streams (EFSA (2006) suggest that such changes may be better accommodated within the more realistic framework of catchment-scale modelling.)	Current FOCUS assumption is that 100% of the catchment area is cropped and that 20% of this area is treated at the same time as the 1 ha field	The fraction of the upland catchment that is treated is a function of regional crop density and the percent of crop treated. The FOCUS assumption of 100% cropped generates excessive estimates of runoff volume.	When larger catchments are considered, crop density is likely to decrease from 100% to peak values of 50-60%, affecting the relative contributions of runoff and base flow. The percent area treated with chemical is a function of both the crop density and the typical marketshare of the chemical.	Use GIS and/or crop statistics to determine typical crop densities in various size catchments. Crop density varies by type of crop as well as by region. Percent crop treated can be determined from local or regional chemical usage data. However, reliable sales data is frequently difficult to obtain, especially for relatively small geographic areas.
Cross section of ditch and stream	Cross section is currently assumed to be rectangular	Actual ditches and streams generally have sloping sides that result in increased width as the depth increases. The rectangular cross-section primarily affects the degree of realism with which the water depth varies with discharge (the Q(h) function) rather than concentrations.	Assume a realistic cross-section including bottom slope and roughness to improve the simulation of Q(h).	Assume a side slope sufficient to ensure stable hydrologic calculations. By its nature, this assumption is likely to be empirical. Changing the slope assumption in TOXSWA 2.0 would require modifications to the existing code.

General considerations: all models

Factor	Value assigned by FOCUS	Issue with FOCUS Value	Factor(s) to change	Source of Data to Support Factor Change
Spatial uniformity	Runoff, erosion, drainage and subsurface flow are determined from a single set of soil, cropping and management parameters	Runoff, erosion and drainage exhibit significant spatial variation as a function of soil type, crop type and land management. Current selections have been performed to create a reasonable worst-case for each scenario.	To support probabilistic spatial modeling, a range of properties could be assigned to key soil, cropping and management properties. For larger catchments, a distribution of appropriate land uses could be used to improve realism.	Typical land use and soil types can be obtained from various local, national and/or EU level databases. Ranges of cropping and management properties are likely to be based on expert judgement.
Temporal uniformity	Concentrations in water bodies are calculated from a single selected climatic year	Runoff, erosion and drainage exhibit significant year-to-year variation in response to changing weather patterns. Longer simulations provide a range of values that can be expressed probabilistically.	Each of the FOCUS scenarios currently has 20 years of data and a single year is selected for use in Step 3. Simulation of all 20 years would require modifications to the current Step 3 models.	Probabilistic evaluation of surface water concentrations should ideally be compared with available monitoring data to help place the results in context.

2.1.2 Step 4 refinements based on field-scale modelling

Refinements to the current FOCUS Step 3 results can be divided into three types of changes:

1. Refinement of current field-scale results by making relatively straightforward changes to individual model parameters, altering chemical properties, application rates or dates or specific environmental parameters which influence the loadings from drift, drainage or runoff or the hydrology of the water bodies

2. Refinement of current field-scale results to incorporate the effects of a risk mitigation measure

3. More complex refinements involve the consideration of larger scale evaluations such as:
    - creation of new field-scale scenarios (based on analysis of major cropping areas in Europe)
    - use of chemical monitoring data (which can provide integrated surface water exposure over large geographic areas)

- application of probabilistic approaches (which can extend modelling temporally and/or spatially)
- development of distributed catchment models (which integrate aquatic loadings over large geographic areas)

Various aspects of each of these approaches are discussed in the following sections on modelling, together with an evaluation of the current state-of-the-art in simulating transport and transformation processes.

2.1.2.1 Key chemical properties subject to refinement

Sorption

In most current environmental fate models (including PRZM and MACRO), sorption is represented as an instantaneous, reversible, equilibrium process with the following relationship between the sorbed and dissolved pesticide concentrations:

$$C_{soil} = K_f \, C_{water}^{\,n}$$

where

C_{soil} = concentration of pesticide in soil phase

K_f = Freundlich partition coefficient

C_{water} = concentration of pesticide in water

n = Freundlich exponent

If. $n = 1$, the Freundlich isotherm reduces to a linear relationship between the two phases. In MACRO, the Freundlich equation has been adapted to represent sorption in both the general soil matrix as well as in macropores. When $n < 1$, the Freundlich equation results in progressively higher sorption values at lower environmental concentrations. Many pesticides are found to have a Freundlich exponent of 0.9 which is the default value recommended for use in FOCUS modelling. When Freundlich sorption values are used for modelling, care should be taken to ensure that the data encompass appropriate ranges of actual soil concentrations. Due to the exponent in the Freundlich equation, the solute concentration is reduced considerably at lower concentration levels, potentially leading to underestimation of actual sorption results at low concentrations. The above description would typically apply to sorption in soil but is also relevant to sorption on suspended sediment, on sediment at the bottom of a water body and on macrophytes in the water column.

In some cases, the assumption of equilibrium between the two phases is not valid. In general, situations involving extended contact times between the mobile and stationary phases (such as normal matrix flow through the soil profile) can be represented adequately using an assumption of equilibrium partitioning. In other situations, such as in macropore flow, the velocity of transport in macropores can be so fast that sorption only takes place to a very limited degree, partly because of the distance between the molecule and the pore wall and partly because of the kinetics of sorption. Similar arguments can be made for pesticide being rapidly transported in surface water.

For some chemicals, equilibrium between the soil and water phases is reached over a period of days rather than hours and it is appropriate to consider representing the kinetics of sorption rather than simply assuming a single equilibrium value.

BOX 1

Sorption: Practical Step 4 refinements within the FOCUS modelling framework

Use of time-dependent sorption values

In PRZM, sorption kinetics are simulated by entering a time series of sorption values which increase with time and permit the model to represent time-dependent sorption in soil. MACRO in FOCUS does not currently have an option for simulation of time-dependent sorption. However, MACRO 5.0 does include this option. The FOCUS groundwater scenarios manual contains additional details on the simulation of time-dependent sorption and it should be recognised that use of this option requires the generation of additional data.

The sorption of pesticides is frequently assumed to be primarily related to the organic carbon or organic matter content of the soil and sorption results are commonly normalised using these factors (e.g. Koc or Kom). However, for some chemicals, sorption to other sites, such as sesquioxides on clay particles, may have greater influence on partitioning. Caution should be used to ensure that the sorptive behaviour of a specific chemical is appropriately characterised.

Adsorption may not be fully reversible. In general adsorption values are used in environmental fate models and full reversibility is assumed. If desorption data indicate very limited reversibility of sorption, this will limit the mobility of the chemical. Column leaching studies can provide an indication of the net effective rates of adsorption and desorption and may be useful to characterise the mobility of compounds that do exhibit distinct irreversible sorption. When desorption rates distinctly differ from sorption rates, multiphasic or "shrinking core" models have been used to characterise a fast sorption process which operates in parallel to one or more processes which are slower due to sorption kinetics or diffusion. However, the mathematics become complex and regulatory use of this concept is likely to remain limited.

<u>Hydrolysis</u>

The hydrolysis of organic chemicals in water is often observed as a first-order reaction given by (Thomann and Muller, 1987):

$$\left(\frac{dC_d}{dt}\right)_{hydrolysis} = -k_T^H \cdot C_d\,,$$

where

k_T^H = the hydrolysis rate constant (h^{-1})

The hydrolysis rate constant (k_T^H) may include contributions from acid- and base-catalysed hydrolysis as well as nucleophilic attack by water (neutral hydrolysis). The following equation explains these possibilities explicitly:

$$k_T^H = k_H \cdot [H^+] + k_{OH} \cdot [OH^-] + k_{H_2O}$$

where:

K_H = the acid catalysed hydrolysis rate constant (mol-1*h-1)

K_{OH} = the base catalysed hydrolysis rates constant (mol-1*h-1)

K_{H2O} = the neutral hydrolysis rates constant (h-1)

Generally, hydrolysis is determined at three pH-values allowing calculation of the three constants shown the equation above. The current FOCUS models use lumped first order degradation rate constants. As a result, the only time that hydrolysis data is directly used in FOCUS modelling is when this mechanism is the single dominant mechanism responsible for transformation in surface water. In this limiting case, the degradation rate in the water phase of a water/sediment study should correspond to the rate of hydrolysis.

Evidence of rapid (often pH dependent) hydrolysis from standard laboratory studies required as a component of physico-chemical characterisation can also be used to help explain behaviour within soil degradation studies. Where hydrolysis appears to be the dominant degradation mechanism this can be used as a possible justification for replacing assumptions that rate of degradation declines with depth (assumption that degradation is mainly biological). See subsequent discussion under heading ‘pH dependency of processes’.

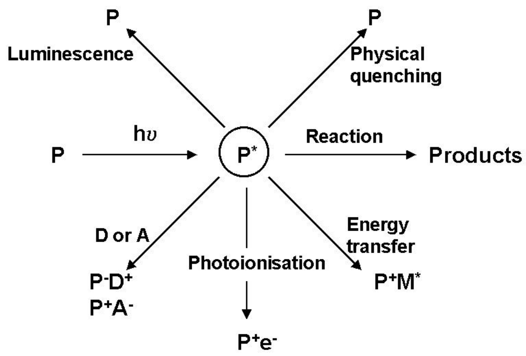
Photolysis

Current regulatory studies of photolysis include experiments in both soil and water. In contrast, aerobic soil metabolism studies as well as water/sediment studies are conducted in the dark. If photolysis is thought to play a significant role in the transformation of a chemical, it is appropriate to ensure that the effects of this mechanism are included in the lumped degradation rates that are used in aquatic fate models such as TOXSWA.

Dissolved pesticide in the water column is subject to photolytic decay, which can be either direct or indirect. Direct photolysis takes place if the chemical absorbs light, and as a consequence, undergoes transformation. Indirect photolysis takes place if the chemical receives energy from another excited species (sensitised photolysis) or very reactive, short-lived species (e.g. peroxy-radicals, singlet oxygen), which are formed due to absorption of light by dissolved organic materials (Schwarzenbach et al 1993). This is illustrated in Figure 2.1-1.

The first-order photolytic decay rate may be calculated as a function of the intensity and the spectral composition of the light and the light attenuation in the water column, which depends on the concentration of suspended matter and on the light absorption spectra of the pesticide.

Figure 2.1-1 Summary of phototransformation processes



To calculate the photolytic degradation one needs to know the quantum yield defined as:

$$\Phi_r(\lambda) = \frac{\text{total number of molecules transformed}}{\text{total number of photons (at wavelength } \lambda\text{) absorbed by the compound}}$$

Assuming that the amount of light absorbed by the chemicals is much less than the amount of light absorbed by the water body the light absorption of the compound per unit volume can be expressed as:

$$\mathrm{I_a}(\lambda) = \frac{\mathrm{W}(\lambda)\cdot\varepsilon(\lambda)\cdot \mathrm{C_d}\cdot\left[1-10^{-\alpha_{\mathrm{D}}(\lambda)\cdot z_{mix}}\right]}{depth\cdot\alpha(\lambda)}$$

where

$\mathrm{I_a}(\lambda)$ = the total number of quanta absorbed of the array of wavelength (λ)

$\mathrm{W}(\lambda)$ = the total light intensity at the surface distributed at the array of wavelength (λ)

$\varepsilon(\lambda)$ = the decadic molar extinction coefficients distributed at the array of wavelength (λ) (mol quant ·m-1)

$\alpha\mathrm{D}(\lambda)$ = the apparent or diffuse attenuation coefficients of river water distributed at the array of wavelength (λ)

Cd = the concentration of active substance

Depth = the depth of the river

Zmix = depth of pond or river

$\alpha(\lambda)$ = the attenuation coefficients of river water distributed at the array of wavelength (λ)

When the total number of quanta absorbed, $I_a(\lambda)$, and the reaction quantum yield, $\phi_r(\lambda)$, are known then a first order photolytic degradation rate, K_{photo}, can be calculated as:

$$K_{photo} = I_a(\lambda)*\phi_r(\lambda), \qquad \text{(Eq. 1)}$$

And the photolytic degradation can then be expressed by the differential equation:

$$dCW/dt = -K_{photo}*CW, \quad \text{(Eq. 2)}$$

Generally, organic compounds including active substances should absorb light in the wavelength range of 290-600 nm in order to be photolytically transformed (Guenzi et al., 1974).

The diffuse or apparent attenuation coefficient is estimated on the basis of, $\alpha(\lambda)$, $D(\lambda)$ and the equation of Neely and Blau (1985):

$$\alpha_D(\lambda) = \alpha(\lambda)*D(\lambda), \qquad \text{(Eq. 3)}$$

where

$D(\lambda)$ denotes the ratio between the average path length and the depth for an array of wavelength (λ)

Neely and Blau (1985) stated that $D(\lambda)$ is between 1.05 and 1.3 for blue and UV light in surface water and Schwarzenbach (1993) stated that $D(\lambda)$ might be 2 in very turbid water.

Assessments of degradation mechanisms in the laboratory are conducted under controlled and standarised conditions and individual modes of degradation such as biotic degradation, abiotic hydrolysis and photolysis can be distinguished and the effects of temperature and moisture can be evaluated. Although it is possible to use these individual degradation mechanisms in fate modelling, the current guideline water/sediment study uses a test system which combines the effects of hydrolysis and microbial degradation but neglects the potential role of direct or indirect photolysis in degradation (i.e. it is not irradiated). When combined with other FOCUS assumptions regarding influences of turbidity, absence of macrophytes in the water column and shading due to riparian vegetation, actual aquatic degradation kinetics may vary considerably from the results obtained from a non-irradiated water/sediment study.

BOX 2

Photolysis: Practical Step 4 refinements within the FOCUS modelling framework

Use of irradiated water/sediment study

If photolysis is thought to play a significant role in the overall degradation of a pesticide, it is appropriate to perform a higher-tier irradiated water/sediment study in order to capture the combined degradation rates of photolysis, hydrolysis and microbial degradation in lumped water/sediment rate constants which can be used in TOXSWA.

Use of soil and water photolysis studies

If a pesticide is readily degraded via photolysis, it may be appropriate to modify the degradation rate on foliage and in the top 1-2 cm in PRZM and MACRO simulations to reflect the role of photolysis in pesticide degradation.

Microbial degradation

In most models, degradation kinetics are based on first order rate expressions in order to describe transformation of pesticides and metabolites in soils, sediment and water. The first order rate equation can be expressed as

$$c=c_o\, e^{-kt}$$

or

$$\frac{-dc}{dt} = kc$$

where

c = concentration of pesticide

k = rate constant

Current FOCUS recommendations require the use of soil degradation rates that have been normalised to standard conditions for both soil moisture and temperature:

$$k = k_{ref} F_W F_T$$

where the soil water content function is given by

$$F_w = \left(\frac{\theta}{\theta_b} \right)^B$$

where B is an empirical exponent (0.7 according to FOCUS recommendations) and θ_b is a reference moisture condition (maximum water-holding capacity according to FOCUS). The soil temperature function is given by a numerical approximation of the Arrhenius equation

$$F_T = e^{0.08(T - T_{ref})}$$

where T and T_{ref} are the actual soil temperature and the reference temperature in ^{o}C.

Degradation rates in water systems are normally corrected only for temperature. At temperatures near freezing, biodegradation is assumed to stop in soil, sediment and water.

Soil degradation studies are performed in moist soils and it is not straightforward to determine the degradation rates associated with each individual phase. As a result, a uniform lumped rate of degradation is usually assumed for both phases in environmental fate models such as PRZM and MACRO.

The FOCUS Kinetics Work Group has recently issued a draft report which provides detailed recommendations for the determining the most appropriate kinetics to use for modelling. Since all current FOCUS models are based on the use of first order rate equations, this guidance document recommends that the best possible single first order (SFO) fit be used, either based directly on a nonlinear first-order regression or an equivalent first-order fit determined from a more complex fit such as first order multiple compartment (FOMC) or double first order parallel (DFOP). This document also provides recommendations for addressing a wide range of experimental conditions, including handling lag phases, compensating for data below detection limits and handling data that vary dramatically from expected values (i.e. outliers). The reader is referred to the FOCUS kinetics document for more details on the recommended calculations for both parent chemicals as well as metabolites (FOCUS Kinetics, 2004).

<u>pH-dependency of processes</u>

Several of the processes mentioned above are characterised by a dependency with pH conditions. To volatilise, molecules have to be in neutral form, meaning that ionised compounds are typically nonvolatile. However, for compounds which have pKa values within the range of ambient pH conditions, the degree of ionisation is a function of both pKa and pH. Sorption is also influenced by the charge of the compound and is therefore pH-dependent for ionised compounds. When degradation is dominated by hydrolytic processes, it is common to observe a dependency upon soil pH conditions. Care should be taken to select degradation parameters that are appropriate to the scenario’s soil pH conditions. Under certain circumstances this may also provide justification for challenging the default assumption that the rate of degradation declines with depth. The pH-dependency of pesticide degradation in soil can be related to the microbiological populations present. It is well known that acid soils have more fungi and less bacteria present (Russell, 1973).

<u>Modelling of metabolites</u>

Pesticide models vary in their capabilities to handle the kinetics of metabolite formation and decline. PRZM is capable of simultaneously simulating a parent compound and two metabolites, generated either in parallel or in series. MACRO handles one chemical at a time. The model is run for the parent compound and an output file is created containing the amount of metabolite formed. This amount is calculated by multiplying the fraction of the compound that is expected to become a metabolite with the amount of compound degraded and the ratio between the molar weight of the metabolite and the original compound. To simulate the fate of the metabolite, MACRO is run again, re-parameterised with the parameters of the metabolite. Currently, TOXSWA can only handle one chemical species per run. As a result, separate TOXSWA simulations are required for the parent and each metabolite, as described in the FOCUS surface water report. As a result of this disparity in the capabilities among the three FOCUS surface water models, the most straightforward way to model metabolites is through the use of approximation which treats the formation and decline of metabolites as the application of an equivalent amount of parent. The "equivalent parent" application rate of metabolite is calculated as follows:

$$ARm = ARp * \frac{F,\max}{100} * \frac{MWm}{MWp}$$

where

Rm = application rate, metabolite (g/ha)

ARp = application rate, parent (g ai/ha)

F,max = maximum percent formed in an environmental compartment (in soil or in the total water/sediment system)

MWm = molecular weight, metabolite (AMU)

MWp = molecular weight, parent (AMU)

It should be noted that the various entry routes of chemical into surface water use different "maximum percent formed" data:

Entry route for metabolite	Data used for "maximum percent formed"
spray drift	max amount formed in water/sediment study
runoff, erosion, drainage	max amount formed in soil study

The FOCUS Kinetics group has provided detailed guidance on analyzing the formation and decline of metabolites in experimental studies and the appropriate use of this kinetic data in modelling. The reader is referred to this document for further recommendations on the modelling of metabolites.

BOX 3

Modelling of metabolites within the FOCUS modelling framework

<u>General recommendation for metabolites</u>

In most cases, the simulation of metabolite formation in the water or sediment layer is only warranted for pond scenarios. The residence time of surface water in the stream (~0.1 d) and ditch (~5d) scenarios is very short and most likely does not permit a significant amount of transformation of parent substance into degradation products. This conclusion does not apply for short-lived compounds which rapidly form metabolites after entering a water body.

The formation and decline of metabolites should be handled as the application of "equivalent parent" (see discussion above).

Case 1: metabolite is formed in BOTH soil AND in aquatic systems (water or sediment)

SWASH – Enter the physical/chemical properties of the metabolite into SWASH. When the parent is applied as a foliar application, it is necessary to estimate the total equivalent metabolite application to soil since neither MACRO or PRZM track foliar formation of metabolites. The resulting equivalent metabolite application should be made as a soil application. Next, the runs for the crop of interest should be defined using the Wizard.

For this case, there are two estimates of the "maximum percent formed", one in soil and one in water. If these two values are similar, a mean value of maximum percent formed should be used to determine the "equivalent parent" application rate used in SWASH. If the maximum percentages are distinctly different, it may be necessary to enter the equivalent parent rate for soil, to run MACRO and PRZM and then to edit the drift value in the TOXSWA input file (*.txw) using the equivalent parent rate based on the maximum percent formed in aquatic systems. In the current FOCUS implementation, TOXSWA will need to be run manually following any modifications to the TOXSWA input file. The procedure for doing this is described in Box 5 below. Finally, SWASH should be used to create the metabolite input files for MACRO, PRZM and TOXSWA.

MACRO, PRZM and TOXSWA

These models should be run using the SWASH shell to generate a TOXSWA output summary file for each metabolite and for each scenario/water body combination. Metabolites which are primarily formed in the water layer can be applied as equivalent parent by means of a drift event. Metabolites which are primarily formed in sediment can be applied to the top 5 cm of the sediment layer as an 'initial' concentration before the start of the simulation.

Case 2: metabolite is formed only in soil

SWASH – Enter the physical/chemical properties of the metabolite into SWASH. When the parent is applied as a foliar application, it is necessary to estimate the total equivalent metabolite application to soil since neither MACRO or PRZM track foliar formation of metabolites. The resulting equivalent metabolite application should be made as a soil application. Next, the runs for the crop of interest should be defined using the Wizard.

For this case, it is necessary to eliminate aquatic loading via spray drift since the metabolite is only formed in soil. This can be done by selecting the application option of "granular application" if the compound is applied to foliage or the soil surface or "incorporated" if the compound is incorporated. Both of these application options result in no drift. If the parent is applied foliarly, the fraction of canopy interception should be used to further correct the "equivalent parent" application rate calculated in Equation 1. The input files should then be written by SWASH for MACRO, PRZM and TOXSWA.

MACRO, PRZM and TOXSWA

These models should be run using the SWASH shell to generate a TOXSWA output summary file for each metabolite and for each scenario/water body combination.

Case 3: metabolite is formed only in aquatic systems (water or sediment)

SWASH – Enter the physical/chemical properties of the metabolite into SWASH. For this case it is necessary to eliminate runoff/erosion and drainage loadings since the metabolite is not formed via these routes. This can be done by entering an extremely short soil half-life (e.g. 0.01 d) for the metabolite. The needed runs should be defined for the crop of interest using the Wizard, the needed applications should be defined using the "equivalent parent" application rate and the metabolite input files for MACRO, PRZM and TOXSWA should be created using SWASH. The simulation of metabolites that are primarily formed in the water or sediment phase is done as described in Case 1.

MACRO, PRZM and TOXSWA

These models should be run using the SWASH shell to generate a TOXSWA output summary file for each metabolite and for each scenario/water body combination.

The process recommended above is conceptually straightforward to understand and relatively easy to organise. However, it is highly repetitive, potentially subject to human error and can take a considerable amount of time, especially when it is necessary to calculate potential aquatic concentrations for a large number of metabolites. Further refinements to more accurately simulate rate of formation within the water/sediment system may be possible through careful modification of macro (*.m2t) or przm (*.p2t) output files (simulation of formation as a series of ‘loadings’ distributed in time).

To facilitate performing Step 3 metabolite calculations as well as Step 4 calculations with reduced drift loadings (e.g. due to increased buffer distances), it would be helpful if

TOXSWA were modified to permit these calculations to be performed automatically within the current SWASH/TOXSWA shell. Relatively simple programming changes in this shell could create the appropriate subdirectories, correctly parameterise the metabolite loadings and calculate the needed water and sediment concentrations. The addition of this improvement to the existing FOCUS surface water models would help ensure a consistent, high quality Step 3 and Step 4 calculation result to support aquatic risk assessment calculations in the EU.

Simulation of formulations

When the pesticide is applied as a spray, it is typically mixed with other components within a dilute formulation. These other compounds are added for a number of reasons including improvement of storage stability, improvement of the rate of dissolution in the spray tank, enhancement of efficacy following application and increasing solubility or plant uptake. In some cases, formulated products may have different product toxicity than the active substance alone. As a result, aquatic effects assessments consider both the active substance alone as well as the formulated product.

Currently, none of the FOCUS models are able to handle formulations explicitly. From an ecotoxicological standpoint, this is of most direct relevance for drift and, to a lesser extent, surface runoff, as the pesticide and the formulation compounds are closely associated just after spraying. Some formulations can have a significant influence on runoff potential (Burgoa and Wauchope, 1995; Wauchope and Leonard, 1980; Leonard, 1990; Wauchope, 1978; Hartley and Graham-Bryce, 1980). Formulations which are designed to control the rate of release into soils (e.g. ‘slow release’ formulations) can influence runoff potential. Such formulations may have the effect of extending the product’s efficacy but, in doing so, may also extend its apparent persistence and availability for runoff. In addition, certain surface applied formulations may be designed to increase availability (and, therefore, efficacy) at the soil’s surface but, in so doing, increase the potential availability for runoff.

It is unlikely that the active ingredient and its formulants will remain associated during transport in the soil because sorption, degradation and dissipation behaviour (including volatility) are likely to differ. These differences may be less marked if transport takes place through the macropores.

It is important to recognise that issues may emerge with certain formulations that cannot be adequately represented by current regulatory models. For example, the potential for transport of solid phase formulations (i.e. granules) into surface waters has been raised as a cause of regulatory concern in the past where the efficiency of incorporation cannot be guaranteed. Such transport cannot be effectively simulated by current regulatory models and must be addressed through more empirical approaches.

BOX 4

Formulations: Practical Step 4 refinements within the FOCUS modelling framework

Controlled release

In Step 4, a controlled release formulation could be simulated kinetically as a parent/daughter pair with the daughter product being the active ingredient that is released into the environment. Alternatively, a single application rate of a controlled release parent compound could be represented as a large number of small individual applications to mimic the effect of the formulation.

Incorporation

The FOCUS Step 3 simulations include the option of using various types of incorporation (see FOCUS Surface Water manual). In the event that none of the available options adequately describe the application of the product, it is possible to manually enter a chemical concentration profile as the time zero concentration in PRZM as a Step 4 refinement of incorporation. The runoff and surface water calculations can then be performed as in Step 3.

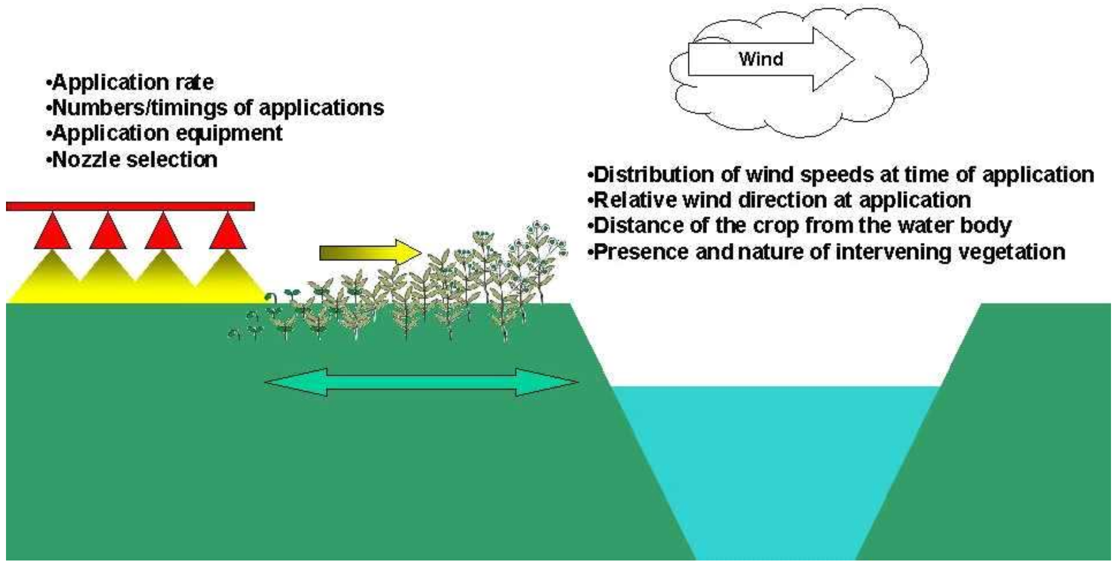
(see also Table 2.1-2)

2.1.2.2 Key transport processes subject to refinement

<u>Spray drift</u>

The critical characteristics that have a significant influence on drift are summarised in Figure 2.1-2.

Figure 2.1-2 Summary of critical influences on spray drift



The spray drift values used in FOCUS surface water assessments are based on ground application drift data compiled by the BBA (Ganzelmeier et al, 1995 and Rautmann et al., 2001) and aerial drift results from the USA AgDrift model (AgDrift, 2000). The results of individual drift trials for various types of applications to crops have been compiled and, for single applications, the 90th percentile values have been selected at each distance. A regression has been developed to fit these experimental data as a function of distance only. For multiple applications, lower percentile drift profiles are combined in order to approximate an overall 90th percentile assessment (e.g. 3 applications assuming 77th percentile drift profiles are considered equivalent, overall, to a 90th percentile assessment). The regressions are based on an exponential relationship of drift on distance from the field boundary or spray nozzle and generally provide a reasonable fit to the experimental data. Additional drift datasets are available (e.g. van den Zande, 2002) and are discussed in more detail in Section 1.3 of this report.

Generally, there are no considerations made for the potential that fields are elevated above the surface of water bodies such as ditches, ponds and streams. However, mitigation measures can be easily implemented if measurement data exist – a new regression equation that describes the conditions with the mitigation measure can be proposed.

A more detailed discussion of the various mechanisms that contribute to spray drift is provided in the section on mitigation in this volume (see Section 1.3). Various aspects of the processes involved in spray drift have been modelled (e.g. Asman et al., 2003), but physical modelling of spray drift is generally not included in most practical simulations of drift.

It has been recognised by the FOCUS Surface Water Working Group that, in some cases, it may be necessary to further refine the Step 3 drift values obtained by considering additional factors which affect drift in "real world settings" such as:

a) actual distances between the treated crop and the surface water bodies;

b) evaluation of the drift-reducing effects of cover crops or weeds in the non-treated zone between the edge of the field and adjacent surface water;

c) consideration of the density of treated fields in a landscape and the range of distances between treated areas and receiving water, typically based on GIS analyses;

d) evaluation of the effects of variable wind speed and direction on drift loadings;

e) evaluation of the effects of drift-reducing nozzles or shielded spray equipment.

Box 5 outlines practical approaches and a series of benchmarks that can be considered when developing a refinement of spray drift loadings to surface water in the existing surface water scenarios. Inputs to the calculator include the application rate, number of applications, type of crop and type of water body.

BOX 5

Drift: Practical Step 4 refinements within the FOCUS modelling framework

A drift calculator is incorporated into the FOCUS SWASH (Surface Water Scenarios Help) shell to assist in the development of Step 4 assessments. Employing this calculator it is possible to manually adjust the distance between the treated crop and water body and to evaluate the resultant drift loadings. Any changes made to the drift calculations should be clearly labelled as Step 4 refinements in subsequent reporting.

Drift loadings with the FOCUS Step 3 scenarios are based upon a set of minimum margin/buffer distances which vary as a function of the type of crop and water body. In order to evaluate alternative drift loadings (based upon the FOCUS drift calculator or other values that the notifier may wish to justify; e.g. for drift reducing nozzles), it is necessary to edit the TOXSWA input file. This is accomplished as follows:

- Use SWASH to create a standard Step 3 input file for TOXSWA (*.txw)
- Create a new subdirectory (e.g. named Step 4) and copy the .txw file, the appropriate TOXSWA met files and the TOXSWA batch file (*.bat) to this location
- Run the drift calculator in SWASH to determine the Step 4 drift value for the desired buffer width (or, alternatively, determine the drift value for the proposed mitigation measure, such as drift reducing nozzles)
- Edit the *.txw file and change the drift value (Note: drift values for streams are multiplied by a factor of 1.2 to account for additional drift loading in the upgradient catchment)
- Run TOXSWA in this new subdirectory by using the batch file

(see also Table 2.1-2)

Dry deposition

Dry deposition is a transport pathway that is not currently included in the FOCUS surface water models. This process is associated with deposition of pesticides volatilised from adjacent fields over a relatively short period after application (e.g. up to 3 weeks). In addition, wet deposition occurs, but this process is thought to be a more significant transport mechanism for pesticides over much longer distances.

It is often argued that the concentrations produced by dry deposition are less significant than those resulting from drift. However, some measurements indicate that the volatile losses of pesticide and subsequent deposition can be considerable for compounds with vapour pressures as low as 0.1 mPa (Smit et al. 1998). The process leading to dry deposition is described by Asman et al. (2003), upon which most of the following text is based. Generally, the process is described as emission, atmospheric diffusion and exchange with the surface, both in the non-spray zone and with the water body. If the concentration in the air is higher than the concentration in the air that is in pseudo-equilibrium with surface residues, the net flux will be downward, and dry deposition occurs. In the opposite case, emission occurs.

A FOCUS Air Work Group is currently reviewing models and approaches to describe various types of atmospheric losses and subsequent deposition. The final recommendations of this group should be used to address the issue of dry deposition.

BOX 6

Dry deposition: Practical Step 4 refinements within the FOCUS modelling framework

Recommendations for the possible inclusion of dry deposition in regulatory evaluations of pesticides are currently being discussed by the FOCUS Air Group and will be finalised when their report is issued later in 2004. In the interim, the inclusion of this mechanism is not necessary in FOCUS calculations.

Foliar interception, dissipation and washoff

Regulatory pesticide models normally assume that sprayed pesticide either reaches the ground or is intercepted by the crop foliage. The fraction on foliage is generally determined by the fraction of the surface covered by the crop canopy. Recommended values of crop canopy as a function of growth stage are provided in the FOCUS surface water and FOCUS groundwater reports. However, crops that are not well represented by FOCUS as well as unique application methods may require modification of the simple spray interception assumptions made in the FOCUS methodology.

The dissipation of pesticides from plant surfaces consists of at least three parallel mechanisms: degradation, volatilization and uptake into the leaf surface. In MACRO and PRZM, all three mechanisms are lumped into a single first order dissipation rate. Other models, such as PEARL, allow specification of each of these mechanisms separately.

In MACRO and PRZM, foliar washoff is modelled as a first order process driven by the amount of precipitation received by the plant canopy.

BOX 7

Foliar dissipation and washoff: Practical Step 4 refinements within the FOCUS modelling framework

Foliar dissipation

For FOCUS modelling, a default foliar half-life of 10 days is recommended. If the chemical is known to have more rapid foliar dissipation than this value, an appropriate experiment should be performed to support the use of a more rapid dissipation rate.

Wash off

In the FOCUS surface water report, a correlation is provided to estimate the wash off rate constant from the aqueous solubility. For highly soluble pesticides, the rate of wash off is relatively high and a significant proportion of the chemical can be removed from the plant canopy by a single rainstorm. As a higher tier study, an appropriate wash off experiment can be performed to refine the wash off rate constant for use in regulatory modelling.

(see also Table 2.1-2)

<u>Runoff generation and erosion</u>

The mathematical characterization of surface runoff has followed different schools of thought within different fields of expertise and geographical locations. Several models (e.g. PRZM, PELMO, CREAMS and GLEAMS) are based on the use of the runoff curve number (RCN) method which was originally developed by the USDA Soil Conservation Service (SCS) to estimate runoff volume and peak discharge for design of soil conservation works and flood control projects.

In the RCN methodology, P is the daily precipitation amount, I_a is the initial abstraction (losses to leaves etc.) and P_e is the net storm rainfall, equal to ($P-I_a$). The depth of runoff, Q, is the residual after subtracting F, the infiltration or water retained in the drainage basin (excluding I_a) from the rainfall P. The potential retention, S, is the value that ($F+I_a$) would reach in a very long storm (Maidment, 1992).

If P_e is the effective storm rainfall equal to ($P-I_a$), the basic assumption in the method is

$$\frac{F}{S} = \frac{Q}{P_e}$$

Assuming that F equals (P_e-Q) and that $I_a = 0.2S$, the expression can be rewritten to

$$Q = \frac{(P - 0.2S)^2}{P + 0.8S}$$

For convenience and to standardise application of this equation, the potential retention is expressed in the form of a dimensionless runoff curve number CN, where

$$CN = \frac{1000}{S + 10}$$

The selection of a curve number depends on soil type (four classes), the land use and the antecedent moisture conditions in the catchment, because a wet catchment reacts with more runoff than a dry one.

The use of the curve number method to generate daily runoff values is a significant extension of this methodology from its origins. As a result, a number of validation and comparison studies have been published for PRZM. In the USA, FIFRA Exposure Model Validation Task Force recently completed a validation exercise for PRZM that included comparison of simulated runoff and erosion with the results of field-scale experiments (FEMVTF, 2000). Model predictions for individual runoff events typically agreed with field data within one order of magnitude and cumulative values (e.g. runoff summed over the study period) agreed within a factor of approximately 3X. The accuracy of runoff and erosion predictions corresponded with the magnitude of the runoff events with the order of magnitude accuracy being associated with small events and improved accuracy resulting from more significant events. A list of the primary conclusions and recommendations from FEMVTF are provided in Table 2.1-3. Wood and Blackburn (1984), using 1600 runoff plots in Nevada, Texas, and New Mexico found differences between observed and computed runoff peaks of greater than ±50 % in 67 % of the results.

Table 2.1-3 Main Issues and Recommendations for PRZM (FEMVTF, 2001)

Evapotranspiration	The evapotranspiration routines in PRZM do not provide reliable estimates of ET. As a result, it is recommended that ET be externally calculated and read by PRZM (this is done in the FOCUS calculations). ET extraction in PRZM occurs over a somewhat arbitrary extraction depth. PRZM cannot account for upward water movement due to ET in the upper soil profile.
Soil Loss (Erosion)	Although some variability is expected between observed and predicted soil loss values due to empirical nature of soil loss equations, the predictions may be improved by a better representation of storm intensity in the soil erosion submodel. Currently the peak runoff rates in the erosion model are derived from generalised regional rainfall distributions. A better representation of the rainfall distribution may be helpful in improving the soil loss predictions for individual events.
Crop Characterisation	PRZM-3 (Version 3.12 and subsequent) allows multiple sets of input values for crop cover (C) and Manning's surface roughness coefficients (N). A more detailed description of C and N factors during the cropping period represents the dynamic nature of crop cover and roughness and improves the sediment loss predictions. A seasonal variation in runoff curve numbers (similar to C and N factors) may be helpful in representing the effects of changing crop growth stages on predicted runoff. Also, further investigations are warranted for determining the source of discrepancies and improving the model predictions for smaller runoff events.
Crop Growth	The actual time and extent of maximum canopy coverage may vary depending on how well the crop is growing. The extent of maximum canopy and time of maximum canopy, in turn affects the interception and therefore pesticide losses with runoff and sediments. The time and extent of maximum canopy cover calculated from measured canopy cover data can improve model predictions for interception and washoff. The maturation date in PRZM input sequence should represent the time of reaching maximum canopy cover for a given crop.
Surface Extraction	The non-uniform extraction model currently used in PRZM3 does not account for seasonal variations in soil condition and texture. For example, a freshly tilled porous soil would have different pesticide and extraction characteristics than a compacted soil.
Site-Specific	Site-specific situations (e.g., a runoff event spanning over a period of multiple days) need to be carefully represented in the simulation by adjusting the available input/output parameters.

Much of the following description of the runoff and erosion capabilities of PRZM has been taken from the FOCUS Surface Water Scenarios Report (FOCUS, 2002).

A number of European studies of runoff and erosion have been published and were consulted during the parameterization of the PRZM runoff scenarios (Lennartz *et al*., 1997; Louchart *et al*., 2001; Voltz *et al*., 1997; Sanchez-Camazano *et al*., 1995; Vicari *et al*., 1999). In runoff studies from no-till and tilled fields in a wine-growing catchment in southern France, seasonal runoff losses ranged between 18-34% of precipitation and resulted in median annual losses of 1.3% of applied diuron (range = 0.7% to 3.3%) and median annual losses of 1.0% of applied simazine (range = 0.5% to 3.0%) (Lennartz *et al*., 1997; Louchart *et al*., 2001). When expressed on a watershed basis, the losses were approximately 0.52% for diuron and 0.24% for simazine. In the experimental studies, essentially all of the chemical transport was via runoff. The FOCUS scenario corresponding to southern France is R4 and the selected meteorological year (based on median hydrological response) is 1984. Based on this scenario, the seasonal runoff losses simulated by PRZM were 30% of precipitation with median annual losses for diuron of 0.5-0.7% (depending upon physical property assumptions) and for simazine were 0.4-0.5% (again, depending upon physical property assumptions). All of the simulated chemical transport was via runoff.

In a hilly area at Ozzano Emilia (Bologna, Italy), plots with a 15% slope on a loamy soil were used to study the effect of two tillage systems, conventional tillage (CT) and minimum tillage (MT), on runoff losses of several herbicides. In the year 1996-97 the fate of metolachlor, atrazine and its metabolites (desethylatrazine: DEA; desisopropylatrazine: DIA), and two sulfonylureas, prosulfuron and triasulfuron, applied to a winter wheat-maize biennial rotation was monitored. Runoff losses ranged between 0.1 to 2% of precipitation. As a consequence of the rainfall pattern, losses of herbicides amounted to a maximum of 0.24, 0.25, 0.05 and 0.003% of the amount applied, for atrazine, metolachlor, prosulfuron and triasulfuron, respectively and the minimum tillage reduced metolachlor and atrazine losses with respect to conventional tillage (Vicari *et al*., 1999). In an earlier but similar experiment, carried out in the year 1991-92 using the herbicides atrazine, metolachlor, terbuthylazine runoff was 3.5 and 0.5 % of precipitation for the minimum and normal tillage respectively. A maximum of 1.6, 1.1 and 0.07 % of the applied amount of metolachlor, atrazine and terbuthylazine respectively were lost by runoff. As for the studies in southern France, when expressed on a watershed basis (273 ha) the losses were reduced by a factor of ten (Rossi Pisa *et al*., 1992). The FOCUS scenario corresponding to Bologna is R3. When runoff losses for the compounds studied were calculated using PRZM, parameterised either from the R3 scenario data or the local field data, they were larger than those measured in the field suggesting that, at least for these compounds, the R3 scenario represents a more conservative assessment of exposure than that measured locally (Miao *et al*., 2001).

These PRZM simulation results indicate that the RCN methodology is capable of providing reasonable estimates of the runoff coefficient (fraction of precipitation resulting in runoff) as well as of cumulative runoff flux. More detailed, site-specific comparisons of individual runoff events would require use of local soil, agronomic and weather data.

A recent compilation of runoff studies has been published by the USGS, covering an extremely wide range of scales (from bench top to major watersheds), physical locations (primarily USA and Europe) and chemicals (Capel *et al*, 2001). Analysis of this data set indicates that the mean runoff losses reported for all scales of European study sites was 0.8% of the applied chemical. For small watersheds similar to those used in the FOCUS scenarios (0.1 to 100 ha), the mean runoff was 0.7% of the applied indicating that runoff losses are essentially independent of the size of the watershed. This result supports the use of FOCUS runoff scenarios as representative of larger land areas that are intensively cropped and treated.

An alternative approach to simulating runoff is to determine the amount of infiltration that occurs and then treating non-infiltrating precipitation as potential runoff. In this approach, infiltration can be described with ”infiltration equations” such as Green and Ampt (1911) or Richards equation. In both cases, the hydraulic conductivity of the soil, the moisture content, and the porosity are important parameters. The generated runoff is thus automatically a function of antecedent moisture. One advantage of the Richards equation approach is that layers with low permeability will limit infiltration and thus influence runoff. Also the choice of lower boundary condition for the soil column may influence the simulated surface runoff for this type of models during wet periods.

A complicating factor with respect to runoff generation is the presence of macropores. The MACRO model (Jarvis, 1991) has a simplified description of surface runoff generation with runoff generated if either the soil saturates to the surface or the saturated hydraulic conductivity of the soil (micropores plus macropores) is exceeded. In practice, surface runoff is only very rarely simulated to occur by MACRO. Other models (i.e. MIKE SHE, Abbott et al., 1986, Refsgaard and Storm, 1995) allow a surplus of water to occur on the surface if the total hydraulic conductivity of the soil is exceeded. The fate of the surplus water then depends upon the type of model. Generally some storage of surface water is allowed. If the storage capacity is exceeded, water is removed either instantaneously, or routed through a kinematic (or diffusive) wave description.

Descriptions of erosion are similarly split up in schools. The runoff-curve based models tend to apply a variation of the Universal Soil Loss Equation for erosion calculations. The current version of PRZM contains three methods to estimate soil erosion: the Modified Universal Soil Loss Equation (MUSLE), developed by Williams (1975); and two recent modifications, MUST and MUSS.

MUSS is specifically designed for small watersheds and was selected for use in FOCUS:

MUSS: $$Xe = 0.79 * (Q * q_p)^{0.65} A^{0.009} K * LS * C * P$$

where Xe = the event soil loss (metric tonnes day^{-1})

Q = volume of daily runoff event (mm)

q_p = peak storm runoff (mm/h), determined from generic storm hydrograph

A = field size (ha)

K = soil erodability factor (dimensionless)

LS = length-slope factor (dimensionless)

C = soil cover factor (dimensionless)

P = conservation practice factor (dimensionless).

This expression depends primarily upon daily runoff volumes and rates as well as the conventional USLE factors K, LS, C and P. It is very weakly dependent on the size of the field.

USLE-based equations are intended to characterise long term ($\geq$10 years) average yearly erosion. Similar to the situation with runoff, the use of these equations for event-based simulations is a significant extension of their original use and, as a result, erosion calculations using PRZM are not likely to predict extreme events. The FIFRA Exposure Model Validation Task Force also included comparison of erosion data with daily simulations and concluded that the USLE routines in PRZM were sufficiently accurate to support regulatory use.

The physically based schools of erosion modelling are based on descriptions of splash erosion, detachment by surface flow in sheet- and rill flow, deposition of particles, sorting and changes in the top layer of the soil. The primary factors which influence erosion are slope, depth of water in the surface layer and surface roughness. Erosion models such as LISEM (http://www.geog.uu.nl/lisem/) and EUROSEM (Morgan et al., 1999) are based on these principles. LISEM is implemented in GIS, while EUROSEM is implemented in a network of planes and channels, thus both allow a description of the surface topography.

The most coherent erosion theory has been developed by Rose and co-workers, and is described in, among others, Hairsine and Rose 1991, 1992a, 1992b, Rose *et al*. (1994) Sander *et al*, 1996, Parlange *et al*., (1999) Beuselinck *et al*. (2002). Furthermore, Ghadiri and Rose

(1991a, b) have investigated issues such as enrichment ratio based on particle sorting principles and found that sorting does not fully account for the enrichment of some nutrients and pesticides seen in transported sediment. Instead they argue that the enrichment is due to the fact that the highest concentration of compounds is on the soil surface or on the outside of soil aggregates and that splash erosion ”peels off” the enriched layers first. However, the model mainly concerns itself with erosion, and the description would have to be combined with other model components to be directly applicable for simulations of transport of pesticide to surface water.

The simple assumption included in the FOCUS Step 3 runoff scenarios is that the treated fields slope uniformly toward adjacent surface water with a maximum slope of 5% or less. Steeper areas were assumed to be terraced to 5% slopes. A landscape approach would include a combination of different slopes, possibly different combinations of soil, vegetation and slope, surface roughness and detention storage on the surface. Furthermore, particularly with respect to erosion, the location of deposition regions can strongly influence the amount of erosion that ends up in the water body. A steep slope followed by a wide flat river valley leads to much less erosion (due to sedimentation) than a landscape with flat topped hills with steep slopes leading to V-shaped valleys. Hill slopes that concentrate flow leads to more serious erosion than landscapes that result in parallel stream lines or even spreads the flow. Due to these effects, erosion loadings into surface water are often very unevenly distributed. The runoff and erosion routines used in FOCUS represent practical implementation of these transport mechanisms and, thus, were recommended for regulatory assessments.

BOX 8

Runoff and erosion: Practical Step 4 refinements within the FOCUS modelling framework

Possible PRZM-specific refinements in runoff and erosion

- Refinement of chemical inputs (which may require additional experimental studies)
- Selection of alternative cropping and application dates
- Selection of simulation year
- Use of probabilistic modelling
- Attenuation of runoff and erosion due to buffer effects
- Evaluation of potential formulation effects

(see also Table 2.1-2)

Details of these refinements are summarised in Table 2.1-2 and are discussed elsewhere in this chapter.

An inherent problem when dealing with runoff generation and erosion at field scale is that the conditions within the field may be influenced by flow from higher parts of the catchment, either through surface or subsurface flow. Secondly, the models generally employed for pesticide registration purposes are single column models, and they therefore represent one slope with a uniform land use, and not the actual land form or the vegetation cover adjacent to the stream. There the processes modelled at field scale are a simplification of the more complex interactions that occur at the catchment scale. As described in section 2.1.3 on catchment modelling, a combination of the present unsaturated zone-processes with a good description of surface runoff and erosion, will require some model development.

Runoff and erosion are, by nature, stochastic events, and serious events occur at irregular intervals. In addition, the events are not generated under "average conditions", but tend to become serious where water accumulates on areas which for some reason are sensitive (due to textural composition, management, topography etc). It is therefore unlikely that one single year will produce events of interest for all application times and that average soil types and slope will represent the worst case conditions. Simulations over several years and perhaps a range of conditions are thus likely to produce more reliable results.

There are a number of relatively straightforward refinements that can be made to refine the runoff and erosion calculations performed by PRZM within the SWASH shell (for critical characteristics, see Figure 1.2). Some of the more straightforward options can be performed within the FOCUS framework such as modifying chemical inputs. However, more complex approaches such as the selection of the simulation year and more subtle aspects such as influences of agricultural practices, soil characteristics, formulations and natural or imposed buffers will require customised modelling. Currently, an ECPA-funded group is developing a simple Step 4 tool to facilitate calculation of higher-tier aquatic exposure values, following the recommendations outlined in the FOCUS Surface Water Scenario and FOCUS Landscape and Mitigation Reports.

As discussed previously, the runoff and erosion routines used in PRZM were originally developed for use in providing longer-term estimates (days to months) of these transport mechanisms. Additional research and model development is needed to accommodate the regulatory need for simulation of aquatic concentrations over short time scales (e.g. minutes to hours).

As summarised in Table 2.1-2 and Box 8, the parameters that can be refined in Step 4 modelling are as follows:

Chemical inputs

The chemical input values that are likely to have the greatest impact on runoff and erosion are the degradation rates in soil and the sorption coefficient. Of these, persistence is likely to be the most important property that influences potential for environmental impact via run-off (Burgoa and Wauchope, 1995). In addition, for foliar applications, the foliar wash-off rates and half-lives may be sensitive. The current FOCUS recommendation is to use geometric mean or median values for chemical inputs. A more complete assessment could be obtained by representing key chemical inputs (such as Koc and soil half-life) as ranges with appropriate distributions and performing probabilistic modelling.

The user has the opportunity of defining alternative application strategies. These may either restrict or manage seasonal timing of applications. Certain application techniques may effectively eliminate (or at least greatly reduce) the potential for run-off (e.g. incorporation of residues entirely to a defined depth). In addition, the PRZM model has the capability of simulating alternative application techniques through careful definition of ‘CAM’ (Chemical Application Method) settings. These are summarised below:

CAM	Description
1	Soil applied, soil incorporation depth of 4 cm, linearly decreasing with depth
2	Interception based on crop canopy, as a straight-line function of crop development; chemical reaching the soil surface is incorporated to 4 cm
3	Interception based on crop canopy, the fraction captured increases exponentially as the crop develops; chemical reaching the soil surface is incorporated to 4 cm – NOTE: This option is not enabled in the FOCUS version of PRZM
4	Soil applied, user-defined incorporation depth (DEPI), uniform with depth
5	Soil applied, user-defined incorporation depth (DEPI), linearly increasing with depth

6. Soil applied, user-defined incorporation depth (DEPI), linearly decreasing with depth

7. Soil applied, T-Band granular application, user-defined incorporation depth (DEPI), use DRFT input variable to define fraction of chemical to be applied in top 2 cm, remainder of chemical will be uniformly incorporated between 2 cm and the user-defined depth - – NOTE: This option is not enabled in the FOCUS version of PRZM

8. Soil applied, chemical incorporated entirely into depth specified by user (DEPI) (modified CAM 1)

9. Linear foliar based on crop canopy, chemical reaching the soil surface incorporated to the depth given by DEPI (modified CAM 2) - NOTE: This option is not enabled in the FOCUS version of PRZM.

Selection of application dates in the FOCUS scenarios

In some cases, it may be difficult to use the Pesticide Application Tool (PAT) in MACRO and PRZM to correctly select appropriate application dates. A herbicide with a single fall application allowed by one or more spring applications can be awkward to handle using PAT. Therefore, it may be necessary to create the appropriate input files using SWASH and then edit the application dates to adjust the selected values as appropriate to represent the actual chemical use pattern. In order to confirm the resulting application dates still fit within the rules used by PAT, it is necessary to either manually check the met files or else to separately run fall and spring applications and use the seasonal dates selected by PAT.

Selection of simulation year

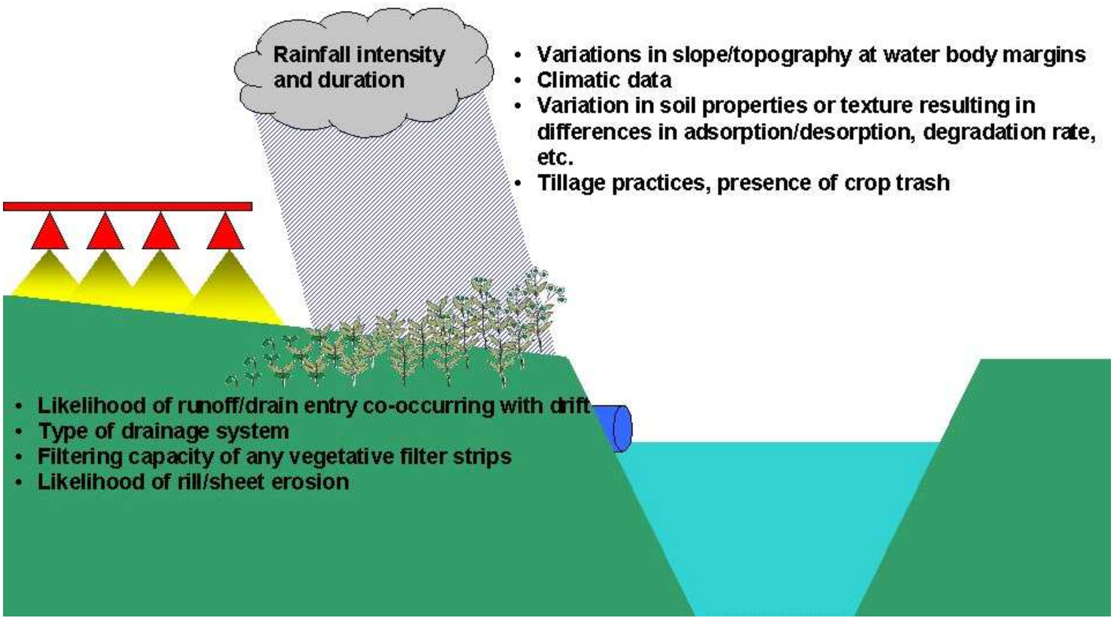
Within the FOCUS Step 3 scenarios a 50th percentile year was selected on the basis of runoff and erosion from two crops; winter wheat (non-irrigated) and maize (irrigated). There are several possible justifications for selection of an alternative representative simulation year such as complex application timing, unique chemical behaviour or distinctly different irrigation patterns than those assumed by FOCUS. Conduct of modelling to consider behaviour and transport potential over a broader time-scale could also be useful by providing probabilistic exposure assessments.

Where long-term weather datasets or statistics are available it may be possible to consider a relatively simplistic approach to help characterise event occurrence potential. The following approach has been proposed by Cohen *et al.* (1995): “Choose one or more intense rainfall events of a known recurrence frequency (e.g. a 2 yr return, 24 h storm event. ‘Apply’ the pesticide shortly before the event and compute the probability of the two events co-occurring and present the probability along with the results.”

<u>Modelling of mitigation measures related to runoff and erosion</u>

Figure 2.2-3 summarises the major factors of relevance in relation to surface losses. Pesticide may move with runoff in soluble form or as erosion. The most common mitigation measure is vegetative strips of different forms (buffer strips grassed waterways), where water may infiltrate and sediment settles. However, all other measures reducing erosion, such as use of cover crops, increased detention storage or efforts to increase infiltration all reduce erosion.

Figure 2.1-3 Major factors influencing runoff and erosion



Simulation of buffer strips (vegetated filter strips)

As discussed in the mitigation section of this report (see Section 1.4), there are several possible methods to mitigate runoff and/or erosion losses from treated fields. In this section, various approaches will be summarised to characterise the efficacy of buffer strips in reducing runoff and erosion. Vegetated buffer strips have been demonstrated to intercept considerable amounts of surface run off (Harris, 1995; Jones, 1993; Patty *et al.*, 1994; SETAC, 1994). There are inevitable questions surrounding the practicality of enforcement of some of these mitigation measures and their applicability within a European and Member State regulatory context is discussed in this report elsewhere (see Section 1.4). However, the recent emergence of remote sensing techniques may provide a means of demonstrating the mitigation potential of vegetated buffer strips within landscape-level aquatic risk assessment exercises.

When considering using buffer zones as a mitigation measure, supporting field studies or existing regulatory benchmarks should be cited to support the proposed buffer efficacy values. When considering evidence from field studies, a useful starting point is the review undertaken by Wauchope et al. (1995) and the studies mentioned in Volume 1 of this report. Within the review by Wauchope, it is pointed out that there are a number of factors that must be considered with care in both the design and interpretation of field studies, including the representivity of:

- Weather conditions
- Cropping practices
- Pesticide application practices
- Tillage practices
- Local agricultural management practices

Benchmark values for the mitigative efficacy of vegetative filter strips have been developed to support national registrations in Germany. These benchmark values are used in conjunction with a surface water assessment model called EXPOSIT (Winkler, 2001). A summary of the selected buffer efficacy values for Germany is provided in Table 2.1-4. Based on these values, the reduction efficiency of any margin width can be estimated using the following empirical relationship:

$$\text{Reduction efficiency (\%)} = 100 - 10^{(-0.083*\text{Margin width} + 2.00)}$$

Table 2.1-4 Summary of vegetative filter strip efficiency proposed by Winkler (2001) for use in Germany

Distance (m)	Vegetative filter strip (*Randstreifenbreite*) reduction efficiency
0	0
5	50
10	90
20	97.5

EXPOSIT assumes that while a reduction in runoff volume reduces the mass of chemical entering the ditch, it also decreases the total volume of water (runoff water + resident ditch water) in which it is diluted in the destination water body. For this reason a 50% reduction in runoff volume may not equate to a 50% reduction in PEC values. As a consequence, if this benchmark approach is employed as a basis for higher-tier modelling at Step 4, it is necessary to consider both a reduction in the mass of chemical and associated volume of water delivered into these destination water body. An example of an EXPOSIT calculation is provided in Table 2.1-5 to illustrate the impact of the volumetric adjustment. **Note that the general principle of a reduction in runoff volume alongside the reduction in chemical loading should also be applied for calculations using the buffer efficacies recommended for use at Annex I (Volume 1, Section 3.5.2).**

Table 2.1-5 EXPOSIT calculation of the efficacy of vegetative filter strips

	Vegetative filter strip width			
Parameter	0 m	5 m	10 m	20 m
VFS efficiency (%)	0 %	50 %	90 %	97.5 %
Volume of run-off water (m^3)	100 m^3	50 m^3	10 m^3	2.5 m^3
Concentration in runoff (µg/l)	3.64 µg/l	3.64 µg/l	3.64 µg/l	3.64 µg/l
Volume of ditch (m^3)	30 m^3	30 m^3	30 m^3	30 m^3
Combined volume (m^3)	130 m^3	80 m^3	40 m^3	32.5 m^3
Adjusted volume to address flowing water conditions* (m^3)	260 m^3	160	80 m^3	65 m^3
Final concentration (µg/l)	1.4 µg/l	1.14 µg/l	0.46 µg/l	0.14 µg/l

* A pragmatic adjustment factor of 2 is used in EXPOSIT to address flowing water conditions

BOX 9

Mitigation of runoff and erosion: Practical Step 4 refinements within the FOCUS modelling framework

Method 1

A relatively simple approach to perform Step 4 modelling of the effects of buffer strips on attenuating runoff and erosion is to post-process the Step 3 *.p2t files created by PRZM for subsequent use by TOXSWA as Step 4 calculations. The primary influence of buffer strips is to reduce the mass loading of chemical entering adjacent surface water bodies via runoff and/or erosion. Viewed mechanistically, buffer zones mitigate runoff by intercepting a portion of the runoff volume as well as sorbing some of the dissolved chemical. Buffer zones reduce erosion losses by intercepting and retaining a portion of the transported sediment which contains bound chemical. **Thus, reductions reported in Volume 1, Table 7 apply both to the volume of runoff water and the loading of dissolved-phase or sediment-bound pesticide in that runoff. Thus, for example, a 60% reduction in dissolved pesticide load will result in a significantly smaller reduction in the predicted environmental concentration because the volume of runoff water (and thus part of the dilution capacity) is also reduced by 60%.**

Before attempting to simulate the effect of buffer zones in mitigation runoff and erosion, it is important to understand that TOXSWA uses the calculated runoff volume reported in *.p2t files to determine the volume of runoff entering the water body both from the treated field and from the appropriate upgradient catchment (only part of which will be treated). As the FOCUS surface water scenarios are abstracted representations of reality, it may be difficult to justify an assumption that vegetated buffers are only applied to part of the upstream catchment; buffers are a long-term investment by a farmer and thus will likely be applied across all fields likely to be cropped with a particular system that requires the buffer. Thus, the most appropriate way to simulate the effects of buffer strips using TOXSWA is to reduce the flux of runoff and/or erosion using the appropriate mitigation factors described elsewhere in this report and to also reduce the runoff volumes calculated at Step 3. This approach will be conservative in all cases because 1) vegetated buffers will seldom be deployed across 100% of an upstream catchment, and 2) some of the runoff water intercepted by the buffer will be routed to surface water. The assumption could be further refined on the basis of landscape analysis and/or evidence of agronomic practice, probably involving the use of a catchment-scale model. In such cases 1) all areas assumed to be treated with the pesticide under consideration should be subjected to the reduction in runoff volumes, and 2) the assessment will need to consider the extent to which runoff intercepted by the buffer is purged of pesticide residues prior to any movement to surface water.

Independent work has been undertaken by ECPA to develop a modelling tool called SWAN. The software operates within the framework of the existing FOCUS surface water scenarios and supports Step 4 calculations through changes to input files for PRZM, FOCUS and TOXSWA. For example, the system allows the user to incorporate mitigation of spray drift or surface runoff or to add in exposure via air where this is known to be a significant route of environmental exposure. SWAN is freely available to users (contact: gerhard.goerlitz@bayercropscience.com).

Method 2

A second method of modelling the efficacy of buffer strips again involves a separate simulation of chemical transport through buffer strips. In this approach, the runoff and erosion values are treated as areal applications to the buffer strip in a second PRZM run. The runoff and erosion amounts generated from the buffer strip can then be read by TOXSWA to create Step 4 results.

Muñoz-Carpena et al. (1999) developed a theoretical description of the hydrology and sediment trapping of a buffer strip. Changes are calculated as the water and sediment runs along the surface of the field and through the buffer strip. Differences in slope and roughness (due to vegetation) are described. The overland flow submodel is based on the kinematic wave approximation, and the infiltration is described by a Green-Ampt equation for unsteady rainfall. The model describes the sedimentation that takes place at the upper end of the buffer strip due to the change in roughness and velocity of the water. This leads to the formation of a wedge, starting already on the upper side of the strip, and after the wedge a zone where suspended load may be trapped. The process is clearly dependent on the particle sizes transported. The model, however, only works with a median sediment particle diameter. Even relatively narrow grass strips remove a considerable amount of sediment. In experiments by Mendez et al.(1999), 4.3 and 8.5 m wide filterstrips removed 82 and 90 % of the sediment load. Most of the sediment is filtered off in the upslope portions of the strip.

The model does not currently include pesticide. In addition to the described model, it would be necessary to include a description of advection/dispersion of pesticide in the runoff, sorption/desorption to particles, and perhaps sorption/desorption to the vegetation. The main issues would be to differentiate between (Mendez et al., 1999):

- Transport of particulate organic material is highly correlated with sediment detachment and transport, and it therefore tends to follow the sediment deposition already described. This is likely to be similar for highly sorbing pesticides,

- Anions that are removed primarily by runoff and infiltration rather than via sorption to eroded sediment, and

- Cations that are removed from runoff by a combination of infiltration and adsorption to the soil, sediment and organic matter in the filter strip. Removal of ammonia-N in the above experiment was 56% and 85% respectively for the two filter strip widths.

Erosion models are commonly event-based models, and little emphasis is given to what happens between events. In this case, however, the degradation of the pesticide in the strip may be of interest, as a long-term accumulation of pesticide in strips near water-bodies could be undesirable. If the USLE equation is used for calculation of erosion, cover crops influence the C-factor, and terracing, contour cultivation and the like, the P-factor in the equation. Infiltration is not easy to deal with in the USLE, but some of the "clones" of the USLE use runoff indicators instead of the rainfall factor. The runoff curve number used for estimating runoff could, however, be affected.

Until a physically based model is available, the effect of buffer strips may at best be described on an empirical basis using one of the two PRZM-based methods covered above.

<u>Unsaturated zone flow</u>

The FOCUS drainage scenarios rely on the MACRO model for description of transport in the unsaturated zone. MACRO is very close to a state-of-the-art model with respect to the description of unsaturated zone flow. The model is based on Richards equation for flow in the soil matrix, similar to several other physically-based root zone models. The description of macropores is comprehensive, allowing exchange between matrix and macropores at all levels. The model allows definition of a range of lower boundary conditions, including field drains. Compared to other root zone models with macropores, there may be differences in the proportion of water that is expected to infiltrate, in the description of exchange between macropores and matrix, and in the exact way drainage is described. However, the concepts implemented are generally well accepted.

Solute transport in the soil matrix is based on the advection-dispersion equation, which is also well established in solute transport modelling. In MACRO, dispersion is set to 0 in the macropores, as convective transport is expected to dominate. As for the water part, there may be differences in the manner in which the exchange of solute between macropores and matrix is described in different models.

<u>Drainflow</u>

Drainage is described as a lower boundary condition to a root zone model – or, in groundwater models as a process in the uppermost layer of the groundwater. The process is generally initiated when groundwater raises above the drain depth and the process is driven by the height difference between the groundwater and the drain, and a drain constant is used to determine the speed with which water enters the drains. In some models, the rate of drainage is controlled by the hydraulic conductivity rather than a drain constant.

It is sometimes observed in fields that drainage occurs before the groundwater rises. These early events may carry rather high concentrations of pesticide. It is thought that this may occur if layers with significantly lower hydraulic conductivity appears near or just under drainage depth. The layers above may saturate and water drain off, although the general groundwater level has not yet responded. Root zone models seldom describe this type of event.

For model simulations at the field scale, a single soil column is generally selected to represent an entire field, meaning that only one set of unsaturated zone parameters and one set of lower boundary conditions represent the field conditions. This may be reasonable for small areas, where the objective is to identify “effective parameters” to represent an area. However, it is quite difficult to simulate ”average” drainage conditions with a single root zone model. In nature, water moves horizontally as well as vertically, meaning that it moves in every direction in which moisture potential gradients are generated. This can result in complex transport behaviour. In a typical root zone model, only vertical flow is allowed in the soil, meaning that it has a tendency of overestimating drainflow and evaporation at high points, and underestimating drainflow and evaporation in low points. The wet soils are usually much more prone to macropore flow, and average conditions may thus underestimate the transport. To catch the differences within a catchment with drained soils, a coupled groundwater/surface water model is required.

Drainage parameters which are subject to refinement in the FOCUS step 3-scenarios

As summarised in Box 10, the number of parameters that are subject to refinement in drainage modelling is significantly smaller than in the case of runoff modelling. Similarly, there are significantly fewer options that can be considered when attempting to introduce mitigation. Issues such as refining chemical inputs and application timing have been discussed in the preceding section. An example of re-evaluation of the site characteristics is given in Volume 1 Appendix 3, and influences of formulations have been discussed elsewhere in this section. The use of probabilistic techniques are briefly discussed in Section 2.1.4. Many of these principles are also applicable to drainage but require an appreciation that these transport mechanisms are driven by processes that occur over different temporal scales. Drainage is generally viewed as a seasonal phenomenon and significant restrictions on application timing are sometimes necessary to introduce credible and effective mitigation potential.

At Step 4, further modelling may be required in order to gain a more complete understanding of the conditions under which drain flow may be expected. It is of critical importance with this route of entry to gain a clear understanding of the duration of drain flow and exposure and how drain flow contributions may be diluted into the destination water body. Depending upon the chemical and crop being evaluated, it may be appropriate to evaluate alternative soils that are known to be associated with the crop of interest (Volume 1, Appendix 3. The interaction with groundwater variations over a catchment as such does require consideration at a step 4-level.

As for runoff, a single year will not represent a particular percentile of risk for all compounds, due to the difference in application time and chemical properties. Macropore flow, for example, will be heavily influenced by the occurrence of rainfall close to application timing, but also by the moisture conditions in the soil at the onset of the event. In order to obtain a broader context it may be appropriate to run the model over a number of years to evaluate the probability of the selected weather year for a specific pesticide. Probabilistic modelling (discussed in Section 2.1.4) or hydrological modelling at catchment scale may be particularly useful in developing a more complete understanding of the context of drain contributions to exposure within the landscape. For further information on drainage behaviour and its interpretation can be found in the following references: Armstrong and Harris, 1996; Armstrong and Jarvis, 1997; Goss et al., 1983; Harris et al., 1991; Harris et al., 1994; Robinson and Armstrong, 1988; Wesström et al., 2001

Mitigation strategies

Because many drained soils are susceptible to cracking and macropore flow there is great potential for rapid transport into sub-surface drainage systems. Field research programmes designed to investigate drainage losses and the influence of macropore transport mechanisms (including the well-known research programmes initiated in the UK at Brimstone Farm – the basis for FOCUS Scenario D2) have considered a variety of mitigation techniques to reduce the environmental impact of drainage. For example Harris (1995) considered restriction of subsurface drainage to periods when the water table was present in the soil profile, thereby reducing the impact of losses to drains through cracking clay soils. There are, however, very few practical and enforceable regulatory mitigation options open to notifiers seeking to reduce exposure potential where drainage is the most significant route of entry. Other approaches include tillage practices that may influence drainage potential as described by Brown et al. (2001) where the authors have pointed out that these and other approaches require further research in order to determine whether or not these represent practicable management options in heavy clay soils. It should be noted that the most recent version of MACRO (Version 5.0) permits simulation of the effects of tillage on drainflow. Mitigation methods may have an impact on product efficacy and should be considered with care.

In Denmark, drained meadows are converted into wetlands by disconnecting the drains before reaching the stream and leading the water to meadows in order to enhance denitrification. The practice is also likely to influence the pesticide content of the drain water as the compounds are moving through both oxidised and anoxic zones as well as through vegetation. The effect of this practice on pesticide transport has, however, not been quantified.

BOX 10

Drainage: Practical Step 4 refinements within the FOCUS modelling framework

Possible MACRO-specific refinements in drainage

Modification of GAP (e.g.reduction of application rate)

Partial treatment

Formulation change

Soil drainage susceptibility

Change in application timing

Site cropping (untreated strips)

Simulation of effects of tillage (e.g. with MACRO 5.0)

Crop rotation

Capture and mitigation of drainage water

Leading drainage water through artificial wetlands

(see also Table 2.1-2)

Note: consideration of the above for risk mitigation will be very compound- and use-specific and subject to discussion with Member State regulatory authorities

Colloidal transport

A potentially significant transport mechanism that is currently not included in the FOCUS surface water assessments is transport of pesticide sorbed to fine sediment and colloids via field drainage. Reported losses of dissolved or entrained particles through drains range between 15 and 3010 kg/ha/year (Øygarden et al, 1997; Brown et al, 1995; Kladivko et al, 1991; Bottcher et al, 1981; Schwab et al, 1977). The total losses of hydrophobic pesticides in two reported studies were between 0.001 and 0.2% of the applied pesticide (Brown et al, 1995; Villholth et al, 2000). Between 6 and 93% of this amount was sediment bound. In field experiments, total losses of applied doses of pendimethalin to drains averaged 0.0013 % for two sampling seasons (Holm et al., 2003).

A quantification of the importance of drains for addition of fine particular material to the streams has shown that the drains on average contribute 29% of the sediment and in single intensive rainfall events up to 70% of the total load to a stream (Kronvang et al, 1997) under Danish conditions.

The 6% loss in the sediment phase found by Villholth et al (2000) was associated with a load of sediment of only 50 g/ha/mm, which amounts to approximately 35 kg/ha/year. Laubel et al (1998) found a loss of 120-440 kg/ha/year on the same site during other periods. The pesticides used in Villholth et al (2000) (prochloraz) and in Brown et al (1995) (trifluralin) had similar sorption capacity (K_{oc} of approximately 10,000). The 78% recovery in the particle phase observed in Brown et al (1995), however, may be overestimated as trifluralin is relatively volatile and hence a significant fraction of the dissolved pesticide may have been lost.

In the study by Holm et al. (2003), 67 drain water samples taken from an experimental area had contents of pendimethalin above the detection limit. For these samples, between 0 and 30 % (on average 10-15 %) of the pendimethalin found in drain water samples was associated with particles larger than 0.7 μm (nominal filter size). Samples taken from two Danish catchments showed pendimethalin contents in the particulate phase, above *ca*. 0.2 μm, of 66 % and 36-46 %.

There was a strong correlation between particle content and pendimethalin concentration at the experimental area. Modelling of the observations from the site indicated that for strongly sorbing compounds, such as pendimethalin (K_{oc} of 10000-18000), particle-facilitated transport would completely dominate the leaching through the unsaturated zone to the drains. Even for less hydrophobic compounds, particle-facilitated transport could still be a very important transport mechanism through the unsaturated zone.

The description of colloid transport, and transport of pesticides with colloids is still at the research stage. In principle, the process has much to do with generation of particles for erosion. It is likely that the occurrence of pesticide-enriched particles in the beginning of an event as found by Ghadiri and Rose (1991, a and b) is due to sorption to the surface of aggregates (also considered the reason for similar peaks observed for colloid transport). There is general agreement that the particles are generated at the surface.

Particles with sorbed pesticide are then transported through macropores into the soil. The matrix acts as a filter and removes most of the particles that enters here. However, when the colloids enter a drainage system or a stream, a considerable part of the pesticide desorbs from the colloids due to the low concentration of colloids in the drainage water. It is therefore not that easy to quantify the importance of the process based on monitoring data alone. Samples should be analysed as quickly as possible after sampling to avoid dissociation. A mechanism of colloidal transport is included in the current MACRO model but there is generally inadequate data for parameterization, especially for regulatory evaluations.

BOX 11

Colloidal transport: Practical Step 4 refinements within the FOCUS modelling framework

Because colloidal transport is generally regarded as still at the research stage, it has not been included in the FOCUS process. Inclusion of this process has the potential to increase the transport of highly sorbing compounds to drains compared to current FOCUS simulations.

Transport in water bodies

The water body considered may be a stream, a ditch or a pond. Streams and ditches can be described with the dynamic wave description, solving vertically integrated equations of conservation of continuity and momentum (the 'Saint Venant´ equations):

$$\frac{\partial Q}{\partial x} + \frac{\partial A}{\partial t} = q$$

$$\frac{\partial Q}{\partial t} + \frac{\partial (\alpha \frac{Q^2}{A})}{\partial x} + gA\frac{\partial h}{\partial x} + \frac{gQ|Q|}{C^2 AR} = 0$$

where,

Q	= discharge
A	= flow area
q	= lateral inflow
h	= stage above datum
C	= Chezy resistance coefficient
R	= hydraulic or resistance radius
α	= momentum distribution coefficient

The equations are based on the following assumptions:

- the water is incompressible and homogeneous, i.e. negligible variation in density,
- the bottom-slope is small, thus the cosine of the angle it makes with the horizontal may be taken as 1,
- the wave lengths are large compared to the water depth. This ensures that the flow everywhere can be regarded as having a direction parallel to the bottom, i.e. vertical accelerations can be neglected and a hydrostatic pressure variation along the vertical can be assumed,
- the flow is sub-critical.

For small streams and ditches, the flow is described as one-dimensional. However, models exist that also deal with supercritical flow and are able to solve the equations in up to three dimensions.

For stagnant water bodies, the vertical distribution of solute in the water column may become more important. Generally, small waterbodies are modelled as a single compartment, assuming instantaneous mixing of the water in the pond. However, immediately after spraying, pesticide reaching the water body may be concentrated in a film on the surface and, if the pond is dominated by macrophytes, the rate of mixing may be reduced.

The one-dimensional (vertically and laterally integrated) equation for the conservation of mass of a substance in solution, i.e. the one-dimensional advection-dispersion equation reads:

$$\frac{\partial AC}{\partial t} + \frac{\partial QC}{\partial x} - \frac{\partial}{\partial x}\left(AD\frac{\partial c}{\partial x}\right) = -AKC + C_2 q$$

where

C	= concentration,
D	= dispersion coefficient,
A	= cross-sectional area,
K	= linear decay coefficient,
C_2	= source/sink concentration, q the lateral inflow,
x	= space coordinate and
t	= time coordinate.

The equation reflects two transport mechanisms, that is the advective (or convective) transport with the mean flow and the dispersive transport due to concentrations gradients.

The main assumptions underlying the advection-dispersion equation are:

- the considered substance is completely mixed over the cross-sections, implying that a source/sink term is considered to mix instantaneously over the cross-section
- the substance is conservative or subject to a first order reaction (linear decay)
- Fick's diffusion law applies, i.e. the dispersive transport is proportional to the concentration gradient.

If instantaneous mixing is assumed for a pond, the above equation does not apply because the assumption of total mixing essentially creates a single compartment. If sediment at the bottom of a pond or a stream is considered, the sediment should be described as at least one and preferably several layers in order to obtain realistic distributions of pesticide in this compartment. Advection and dispersion then takes place into the pore water of these layers.

BOX 12

Transport in water bodies: Practical Step 4 refinements within the FOCUS modelling framework

Key assumptions subject to refinement

Catchment characteristics

area, percent treated, hydrology, spatial distribution of treated fields, temporal distribution of catchment and edge-of-field loadings

(see also Table 2.1-1)

The upstream catchment, its cropping and pesticide use are all highly simplified within the Step 3 scenarios. Modification of the assumptions may be a simple refinement at Step 4 where they are shown to deviate markedly from reality (e.g. for applications to specialised crops). EFSA (2006) suggest that such changes may be better accommodated within the more realistic framework of catchment-scale modelling.

The key refinements in simulating transport in the water bodies are primarily focused on adding realism to the upgradient catchment in the stream scenarios. The Step 3 catchment assumptions are very conservative and typically result in peak aquatic concentrations that can be significantly higher than typical monitoring results in catchments larger than 100 ha. Reasonable refinements in the catchment dimensions, spatial distribution of treated fields and temporal distribution of loading events will create more realistic concentration profiles in streams.

2.1.3 Catchment scale modelling

The FOCUS surface water scenarios represent an agricultural field using a single combination of soil, weather and boundary conditions. The simulated area is assumed to have a single crop grown on it and all spraying takes place simultaneously. The upper catchment of the stream is assumed to be hydrologically equal to the column modelled, but 80 % of the upstream catchment is assumed to be unsprayed. In reality, the conditions within a catchment differ spatially, especially as larger scales are considered. Several soil types may be present, topographic variation can influence surface flow patterns, chemical applications can occur at various times and the exposure of the stream can vary due to surrounding vegetation and so forth. To capture this spatial and temporal variation in more detail, a catchment model is required.

As a part on an ongoing research project in the UK, a review of various models was performed to assess their suitability for use in catchment modelling of runoff (White et al., 2003). As shown in Table 2.1.3, the authors of this review concluded that potential models for use in catchment modelling can be divided into three groups: one dimensional leaching models which lack the capability of simulating surface processes, field-scale models which simulate runoff but have limited capabilities of simulating flow routing or spatial heterogeneity and finally, various types of catchment models which simulate both surface processes as well as spatial heterogeneity. Further discussion in this section will focus on describing examples of this third class of catchment models.

Table 2.1.3 Classes of hydrological models considered in the TERRACE review

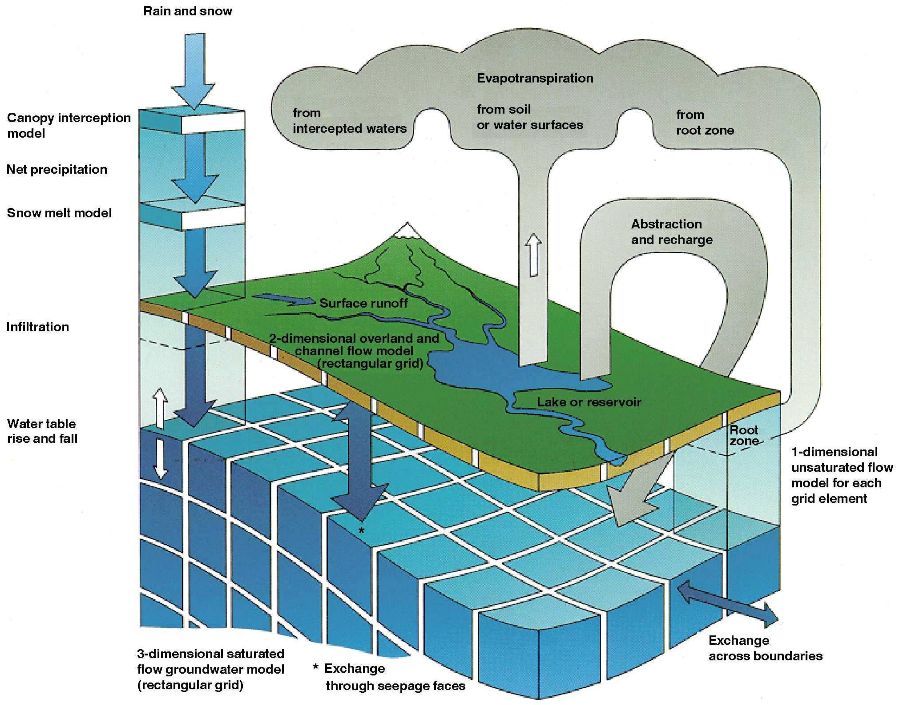
Type of Model	Examples	Potential Use for Catchment Modelling
One-dimensional ("unit area") soil column leaching models	CHAIN_2D CMLS CRACK-NP LEACHM MACRO N3DADE PESTLA PESTRAS PEARL SLIM TETrans VarLeach	No. All of these models lack the capability of simulating surface processes (e.g. runoff and canopy interception)
Field-scale models of hydrological flow, and nutrient and/or pesticide fate	CREAMS EPIC GLEAMS Opus PELMO PRZM RZWQM	No. All of these models are limited to field-scale simulations and do not provide representation of flow routing to low order streams and ditches. In addition, they do not provide adequate representation of spatial variability typically present in catchments.
Catchment-scale models of hydrological flow and nutrient and/or pesticide fate.	AGNPS ANSWERS-2000 HSPF MIKE-SHE SWAT SWATCATCH	All models include capability of flow routing and spatial heterogeneity. Recommended models for further development within TERRACE project include: SWATCATCH , a relatively simple empirical model SWAT, a conceptual model, recommended for use as the default ANSWERS-2000, a complex physically-based model

<u>Short description of the structure of and input to a catchment model</u>

In the field of hydrology, distributed modelling and catchment models have been used for a considerable time. The models used differ in complexity in the description of different hydrological compartments. If solute transport is included, the level of complexity of the process descriptions differs too.

The MIKE SHE model (Figure 2.1-4) is an example of a catchment modelling system and will be used in the following to exemplify the issues of relevance in catchment modelling.

Figure 2.1- 4 Schematic description of the catchment model, MIKE SHE



A catchment model includes a description of the root zone. The description resembles that of PRZM or MACRO (depending on the type of model). However, several soil columns are parameterised to describe the distribution of soil properties over an area. Similarly, climatic data and land use information is given a spatial distribution. Soil properties are usually measured at given points, and usually soil maps are used to generalise the point measurements. Newer techniques such as EM38, providing soil maps on the basis of geo-electrical measurements improve the ability to extrapolate from point measurements to a spatial distribution. A soil type is also characterised by its content of organic matter and its moisture retention properties. For pesticide modelling, values related to sorption and degradation of the compound therefore becomes distributed together with the distribution of the soil properties.

To generalise climatic information, Thiessen polygons or isohyet lines may be used to determine the spatial extent for a particular measuring station.

For vegetation, the distribution in space can be determined from maps, air photos and satellite information or based on statistics.

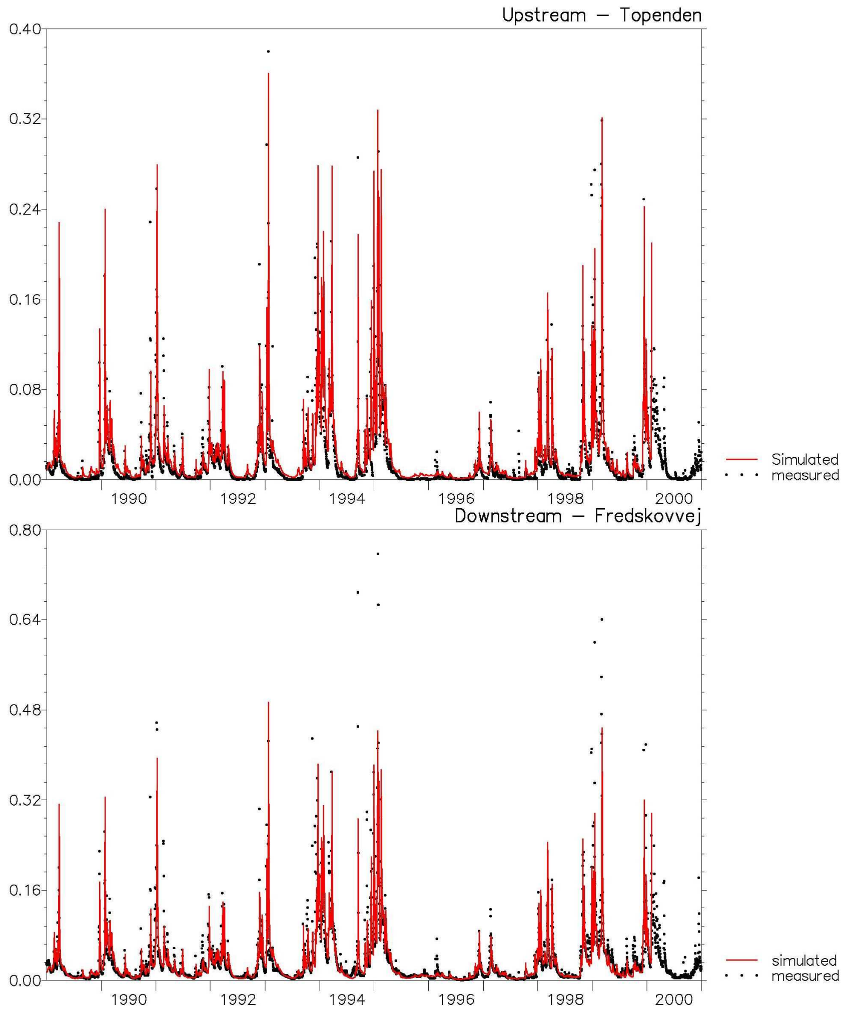
One major difference between the FOCUS surface water scenarios and catchment modelling is that a catchment model is linked to a groundwater model, meaning that groundwater is explicitly modelled. In the FOCUS-scenarios, groundwater flow is mimicked in a very simple way. Catchment models may, however, include groundwater through a full 3-D description or as a number of linear reservoirs. The linear reservoir description interprets the groundwater as a ”bathtub” where water from different parts of the catchment is mixed, and this description is therefore not appropriate for detailed solute transport.

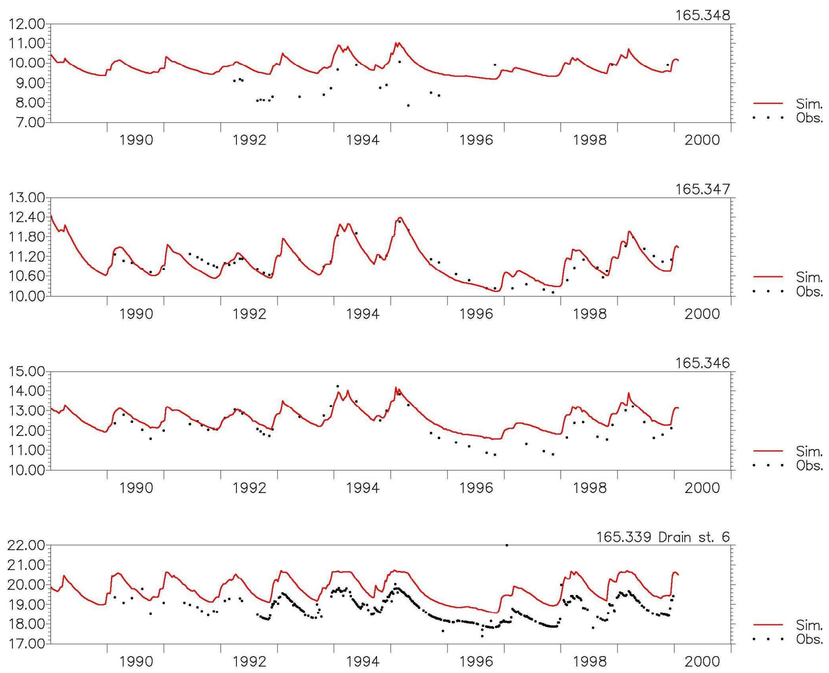
For the models with a 3-D representation, the critical issue may still be the size of the computational units. Where the unsaturated zone-models tend to work with computational layers of maximum 10 cm, groundwater models are working with layers of several meters. Simulated peaks thus tend to flatten out rather fast due to dilution. Particularly in the upper part of the groundwater, generating drain flow, the discretisation of the calculation layers thus has to be done with caution.

A groundwater model is built upon the basis of available geological information from boreholes, geophysical investigations etc. Also for this type of information, it is a key issue to distribute point information in space, and geo-electric measurements may assist in generalising borehole information to profile or 3-D geological models. In areas with important interactions between surface water and groundwater (which is often the case near surface water bodies), the dynamics of the unsaturated zone are significantly influenced by the level of variations in groundwater.

The catchment is linked to a river model that describes the flow in the stream from inside the catchment to the outlet. Water is routed through the system from calculation point to calculation point, according to the cross sections of the stream, the roughness and the slope. Figure 2.1-5 shows an example of a hydrograph generated with MIKE SHE for a small Danish catchment. In this particular catchment, drain flow, groundwater levels distributed over the catchment and stream flow are all well matched.

Figure 2.1- 5 Measured and simulated flow (m^3/s) in the upstream and downstream measuring stations in the sandy clay catchment





Major differences between a catchment model and the single-column models

The main differences between the present scenarios and a catchment model would be

- the distribution of climate, vegetation and soil properties over the catchment,
- the possibility to distribute input such as location of fields, time of spraying and spray drift. Spray drift could, for example, be modified according to occurrence of natural buffer zones in the landscape and exposure of the waterbody due to wind direction,
- the varying conditions with respect to the lower boundary of the soil columns. As the upper part of the groundwater model produces the boundary condition or the columns above, the conditions are influenced by the general topography. Water runs horizontally between grid points in the groundwater model, and may accumulate in lower areas. This particular factor leads to considerable differences in macropore flow between different root zone columns, as macropore flow is induced more often in the wetter areas. Drainage is also selectively activated according to where the groundwater rises above drain level.

In short, the modelling approach catches more variation within a catchment, meaning that some areas will be less vulnerable to transport processes but others may be more vulnerable that a single column approach will reveal.

With respect to the different transport pathways, the following issues may be considered:

Drift is described by the same empirical equations at this level. It is, however, relevant to consider the assumptions further. The stream in the catchment may wind its way through the catchment. The drift received by the stream per square meter, however, should be corrected to ensure that the maximum value, equal to drift perpendicular to the field boundary, is not exceeded. Other considerations at this level are:

a) How large a part of the catchment should be sprayed at the same time.

b) As it physically takes time to run a tractor over a larger area, how large a window should the spray drift be divided over.

c) Should existing (natural or non-natural) buffer zones be kept in the modelling exercise? and if yes, how to incorporate the variation in degree of protection over the catchment.

Dry deposition becomes more relevant at the catchment level, not least as larger areas are sprayed with a particular compound. The general issues are as described in Section 2.1.2.2, but at this scale, it is necessary to consider how to deal with a patchy spraying pattern, if it is not assumed that most of the catchment is sprayed. In addition, the emission and transport processes do not respect catchment boundaries, and the area defined as “relevant for dry deposition calculations” may therefore be somewhat arbitrarily determined, if no natural border such as a forest provides a border.

Runoff generation and erosion: At the catchment scale, it is natural to move away from the single column-models towards a distributed description of runoff generation and movement of water and sediment along the soil surface. This allows the inclusion of topographic features in a much more realistic manner than in the existing FOCUS step 3-simulations. This refinement will not address all the limitations affecting runoff, since models do not currently exist that include all relevant process descriptions related to pesticide transport with runoff and erosion. Thus, although some processes can be dealt with in a deterministic manner at this scale, the effect of mitigation measures such as buffer strips may still have to be handled at least partly empirically.

Issues of scale are highlighted with erosion as assessments increase to the catchment scale. While the catchment may be measured in square kilometres and the field in hectares, the detailed simulation of flow through a buffer strip may require calculation points spaced no further apart than e.g. 0.5 m. Similarly, the demands on the topographic resolutions may be high if actual flow patterns and water depths are to be simulated. The EUROSEM model handles this through assemblies of planes of channels for which the size is defined individually. Most other approaches use calculation units of a fixed size, which then either results in difficulties in disaggregating features or in high computational times.

Colloid transport: Holm et al.(2003) produced a colloid model which was introduced into a catchment model for pesticide simulation (Styczen et al. 2004a). While it was possible to calibrate the colloid generation at plot scale, rather strange results were generated at the catchment scale, because the catchment model also allows the particles to be transported across the surface. It is very obvious that colloid models should be more closely linked with erosion models to yield sensible results when the colloid generation is scaled up, both with respect to effects of crop cover, filtering of particles along the soil surface and deposition. Secondly, it was clear that more work is required to describe the generation of particles, and possibly the enrichment process, in order to produce good results.

Issues, such as that the transported colloids may contain more than average organic material may also be important for the final transport calculation.

Transformation process descriptions act as sink or source terms at each calculation point in the model, similarly to calculations at field scale. The descriptions themselves remain the same. An issue of particular concern at this scale is, however, how the processes should be parameterised in the groundwater. Due to the fact that pesticides may be characterised by greater residence times in groundwater , it is quite important that persistence and sorption behaviour within the saturated zone is defined accurately, otherwise significant under- or over-estimates may result. It makes a significant difference for the concentrations simulated downstream in the catchment, particularly during low-flow situations as these situations tend to be significantly influenced by groundwater. Presently the FOCUS recommendation is that no degradation takes place below a depth of one meter. Research evidence gives a mixed picture. There is evidence that some pesticides degrade more rapidly under oxidised conditions and others under reduced conditions. In order to perform higher-tier catchment modelling including fate in groundwater, it is appropriate to obtain experimental data for the rate of degradation in subsoils and/or groundwater. As sorption usually is modelled as a function of organic content, the sorption potential at depth is usually relatively low.

Intermediate Complexity Catchment Modelling Techniques

As discussed earlier within this report there are a number of assumptions embedded in the FOCUS Surface Water modelling framework regarding how the upstream catchment is recognised and its role in edge of field assessments is simulated. These are necessarily somewhat simplistic in nature and, as a consequence, it may be necessary to consider more sophisticated modelling techniques. The value of catchment modelling in developing a more complete understanding of hydrology and chemical behaviour within a catchment is illustrated in a case study (Volume 1 Appendix 2). Within this case study a thorough characterisation of the catchment enabled sophisticated modelling of chemical behaviour and exposure to be conducted employing the Danish PestSurf model. More information on sophisticated modelling techniques like these and the issues associated with their use is provided later within this chapter. While these modelling approaches are hydrologically highly robust, they unsurprisingly require extensive datasets in order to adequately parameterise a given catchment prior to simulation. At present such detailed datasets are available for a relatively small group of catchments supporting field research activities. Unfortunately, data on a suitable catchment to support simulations of a specific crop and use may not be readily available. Further, it may not be considered cost-effective to develop the detailed catchment characterisation required to support such simulations. As a consequence, it may be necessary to consider less demanding modelling techniques that may, nonetheless, greatly assist in more adequately representing behaviour at a larger scale. The purpose of this section is to provide examples of the range of tools that are currently available and how these are currently employed within resource management and/or risk assessment schemes.

Intermediate approaches can be employed to address the heterogeneity in land use, soils, and pesticide use in a watershed system. Importantly, these approaches can add value to Step 4 in addressing variability in landscape composition and hydrologic attenuation in receiving water systems either within a region or across regions and to provide for a more realistic representation of the spatial or temporal processes that are embodied in Step 3.

Resource management tools

Certain tools remain proprietary with rights owned by companies or government agencies and are, therefore, not publicly available. This fact restricts the potential use of these models for regulatory but discussion of these models can still serve to illustrate principles of catchment modelling. Two examples are discussed here that are in current use within the United Kingdom: CATCHIS and POPPIE. Both of these models were developed to predict concentrations of pesticides in drinking water that is abstracted from surface water sources. These models would need to be applied at a smaller scale to generate results appropriate for use in ecological risk assessment.

Intermediate Complexity Catchment Modelling - Example 1: CATCHIS

The CATCHIS model is an example of a tool developed for the purposes of characterising local risks to water resources to assist in the development of monitoring and management strategies. CATCHIS was developed by the UK Soil Survey and Land Research Centre (now the National Soil Resources Institute) and Severn Trent Water Ltd. With initial funding provided by the UK Ministry and Agriculture, Fisheries and Food (now the Department of Environment, Food and Rural Affairs). CATCHIS is comprised of database and modelling components described in brief here. Databases incorporated into the model characterise soil characteristics, hydrological networks, various landscape features (roads, railways, settlements), cropping and chemical characteristics (including typical application rates and timings) and surface water and groundwater abstraction points. Both surface water and groundwater risk assessments are conducted using models which integrate seasonally dynamic factors relating pesticide usage, land management and weather, with intrinsic but spatially variable factors relating to soil, hydrogeological and hydrological characteristics. The models, called SWAT (Surface Water ATtenuation) and AQAT (Aquifer ATtenuation) have been described by Hollis (1991) and Brown and Hollis (1996). They are based upon the attenuation factor concept developed by Rao *et al.* (1985) and Leonard and Knisel (1988) and also utilise the direct, empirically-derived link between soil types and stream flow established within the development of the HOST scheme (Boorman and Hollis, 1990). The CATCHIS modelling framework has been used to consider relative risks that local water resources will exceed water quality standards (typically the EU drinking water quality standard, but this could be employed in comparisons with ecological/ecotoxicological critera). The resulting spatial risk assessments enable water resource monitoring to be effectively targeted in order to develop protective strategies focusing upon areas where there is likely to be the greatest impact.

<u>Intermediate Complexity Catchment Modelling - Example 2: POPPIE</u>

The POPPIE (Prediction of Pesticide Pollution in the Environment) system is a GIS-based catchment scale model developed by the UK Environment Agency for investigation and prediction of agricultural pesticide concentrations in rivers and groundwaters across England and Wales. POPPIE is used to define and effectively target the Agency’s pesticide monitoring programme. In addition to mapping predicted environmental concentrations of pesticides in the environment, the system is also used to display data on cropping patterns and pesticide usage across England and Wales. The POPPIE system employs a database of agricultural, hydrological, hydrogeological, soil and chemical data similar to that employed within CATCHIS. The surface water modelling framework embedded in POPPIE (SWATCATCH) is also similar to the SWAT model embedded in CATCHIS. The SWATCATCH model was developed by the SSLRC (Hollis *et al.,* 1996; Brown and Hollis, 1996). This is a semi-empirical, distributed model based upon the calculation of flows and pesticide concentrations contributed by each soil hydrological type within a specific catchment. The performance of the model has been assessed in a validation exercise comparing simulations of frequency of detections, maximum concentrations and time series of exposure versus monitoring data in a set of 29 catchments and further information on this exercise is provided by Brown *et al*. (2001).

Building upon experiences with catchment assessments required by legislation in the United States

The U.S. Environmental Protection Agency (USEPA) has been exploring intermediate approaches toward basin-scale (catchment-scale) modelling (Parker et al., 2004). Their study has involved a comparison of three models configured to represent a 242-km^2 catchment area of the White River in Indiana. The evaluation was part of an effort to find tools for carrying out assessments of pesticide levels in drinking water to be conducted under the Food Quality Protection Act (FQPA). The results of the evaluation may also guide the Agency in identifying computer simulation tools that can be used in aquatic ecological assessment for under the Federal Insecticide Fungicide Rodenticide Act (FIFRA). Models selected for evaluation were:

- the SWAT (Soil Water Assessment Tool) model designed and developed by U.S. Department of Agriculture (Neitsch et. al, 2002)
- the NPSM (Non-Point Source Model) component of HSPF (Hydrologic Simulation Program - Fortran) in the BASINS (Better Assessment Science Integrating Point and Non-Point Sources) modelling shell designed by USEPA and
- the RIVWQ (Water Quality Model for Riverine Environments) model designed by Waterborne Environmental, Inc. (Williams *et al*., 1999).

A summary of two of these modelling techniques (SWAT and RIVWQ) is provided, in brief, below:

SWAT

The SWAT (Soil and Water Assessment Tool) model has been developed by USDA to aid in assessing the effect of management decisions on water, sediment, nutrient and pesticide yields with reasonable accuracy on large, ungaged river basins. (Arnold et al., 1998, 1992; Arnold and Allen, 1992; Srinivasan and Arnold, 1994; Srinivasan et al., 1995). This should not be confused with the Surface Water ATtenuation SWAT model discussed earlier in the description of CATCHIS. The USDA model is a physically-based, spatially-related model that requires information about weather, soil properties, topography, natural vegetation, and cropping practices assembled within a customised ArcView Interface. One weather station could be used to represent the watershed or multiple weather stations could be established to provide greater spatial representation. Similarly, within each sub-basin, SWAT allows hydrologic response units (HRUs) to be defined. HRUs are sets of disconnected units in a sub-basin with the same landuse and soil. Algorithms governing movement of soluble and sorbed forms of pesticide from land areas to the stream network were taken from EPIC (Williams, 1995). SWAT incorporates a simple mass balance developed by Chapra (1997) to model the transformation and transport of pesticides in streams. The model assumes a well-mixed layer of water overlying a homogenous sediment layer. Only one pesticide can be routed through the stream network in a given simulation.

RIVWQ

The RIVWQ configuration is actually a hybrid approach that links multiple unit-area simulations of the Pesticide Root Zone Model, PRZM (Carsel et al., 1998) to account for variations in land use, soil and weather across the watershed with an advection-dispersion model to address chemical fate and transport in the receiving water system. Similar model linkages using models preferred by FOCUS is also feasible. Modelling approaches such as these can be configured in relatively detailed or coarse resolution. The watershed can be represented by several predominant land use-soil combinations or as a complex heterogeneous system of many soil and land use categories. Drift can be represented as a constant amount for each simulated pesticide application or weather dependent according to the level of complexity desired. One weather station could be used to represent the watershed or multiple weather stations could be established to provide greater spatial representation. Base flow can be represented as a unit-area contribution or calculated from predicted infiltration and attenuated return flow.

<u>General considerations in catchment modelling</u>

The appropriate level of complexity is a balance between precision in predicting concentrations and resource constraints (time, money, data, and technology). More sophisticated simulations may be provided with the necessity of employing more extensive and higher quality datasets. Consequently, attention should be paid to assessing the primary issue of concern (potential for exceedance of an exposure threshold, return period of an exposure threshold, duration of exposure etc…). Attention should be directed to representing the most sensitive and relevant governing factors. For example, for many compounds, subsurface transport to aquatic systems is insignificant and can be neglected. For other assessments accuracy in exposure duration is more important than accuracy in magnitude. The modeler should attempt to achieve the optimum compromise in level of detail/cost with useful information that will contribute to weight of evidence for making sound regulatory decisions (Williams, pers comm.). Stakeholders can decide whether continued uncertainty warrants additional detail and further investigation, potentially employing more sophisticated modelling techniques.

In future, it is certain that catchment modelling approaches will become a higher profile tool within regulatory risk assessments required under the Water Framework Directive. Therefore, it is likely that this field will evolve rapidly and new and more extensive databases will become available to employing within spatially related modelling frameworks.

More complex catchment modelling – an example used for pesticide registration in Denmark

The Danish EPA has funded an attempt to produce a catchment model for pesticide registration purposes (Styczen et al, 2004, a, b,c). The final product of the project ”Pesticides in Surface Water” is a model tool (PestSurf) that can be used in the registration procedure for new pesticides. PestSurf is based on models of two existing catchments. It is assumed that the two basic models represents some well known Danish conditions. On selected points the basic models are modified to make them more appropriate as general risk analysis tools.

The scenarios are build into a user interface that guides the transfer of pesticide data, the choice of crop, the time of spraying and dose and the width of the bufferzone from the interface to the mathematical models. To reduce simulation time, all the water calculations are carried out in advance and cannot be changed by the user. To the general user of a catchment model for registration purposes, the application of the model would resemble the use of the FOCUS-models, where all general parameters are also fixed in advance.

It is considered close to impossible to create virtual catchments that will represent given percentiles of risk. Rather than producing a complicated set of assumptions, it was considered simpler to work with ”the real world”. One important difference to the existing FOCUS-scenarios is, thus, that the catchments exist in reality and that the model performance can be investigated. It is even more difficult to establish a “risk percentile” for a given catchment, taking into account all transport pathways. Also the issue of similarity becomes complicated for catchments – while two fields may be treated similarly and have the same soil type, two catchments are almost certain to differ in some aspects of land use, soil, geology or climate. In the following, the PestSurf scenarios and their assumptions are described a bit further.

Two small catchments were selected (approximately 4.5 and 11 km^2). It was expected that the upstream part of catchments would be most affected by pesticide use, but no attempts were done to place a percentile on the selected areas. The two catchments are part of a national monitoring system and data regarding climate, land use, farmers practices etc have been collected since 1990. The programme was extended with extra pesticide measurements in order to gather the additional data required.

The hydrological models were first set up and calibrated based on climatic data, stream flow measurements and groundwater levels for a 10-year period. The catchments were modelled in 50 *50 m calculation cells, which are adequate to catch the absolute majority of the hydrological variations, and reasonable also for description of fields. Drift was modelled ”outside” the hydrological model, the effect of buffer strips on drift was taken into account, also if the buffer strips were smaller than 50 m.

From this point, the development followed two paths. An effort was made to model the ”real” situation regarding pesticide application and occurrences in surface water as they appeared in 1999-2000. This exercise was an attempt to investigate the quality of the simulations and was called ”calibration”. Secondly, the conditions for the registration model and its scenarios were defined and set up.

PestSurf (the registration model) includes drift, dry deposition, sorption, degradation and colloidal transport. In the surface water, the compound can undergo hydrolysis, photolysis, biological breakdown or be sorbed to sediment or macrophytes. Transport along the surface with erosion was negligible in the two years, where extended monitoring took place, and this process was therefore left out of the registration model.

Drift is included in the model using empirical relationships as in the FOCUS surface water scenarios. However, although measurements were taken intensively during the spraying season, no drift events were identified. The monitoring programme did thus not quite support the drift calculations.

The registration model is run over 8 years, but the first 4 are considered warming up. The years were selected such that a very wet winter period (2 calendar years) is followed by an average and a dry year. Each series contains two wet springs and two wet autumns. The overall water balance for the four years is very close to the water balance for the 1990’s as a whole. In order to describe macropore-events, 6-minute intensities of rainfall were used in the model. Unfortunately these were not available on site, so data from a station with a similar pattern had to be used. While the monthly rainfall is identical to the local station, the timing of peaks may differ. This is not important for the registration model but this plays a role when simulated and measured pesticide occurrences are compared in the calibration exercise.

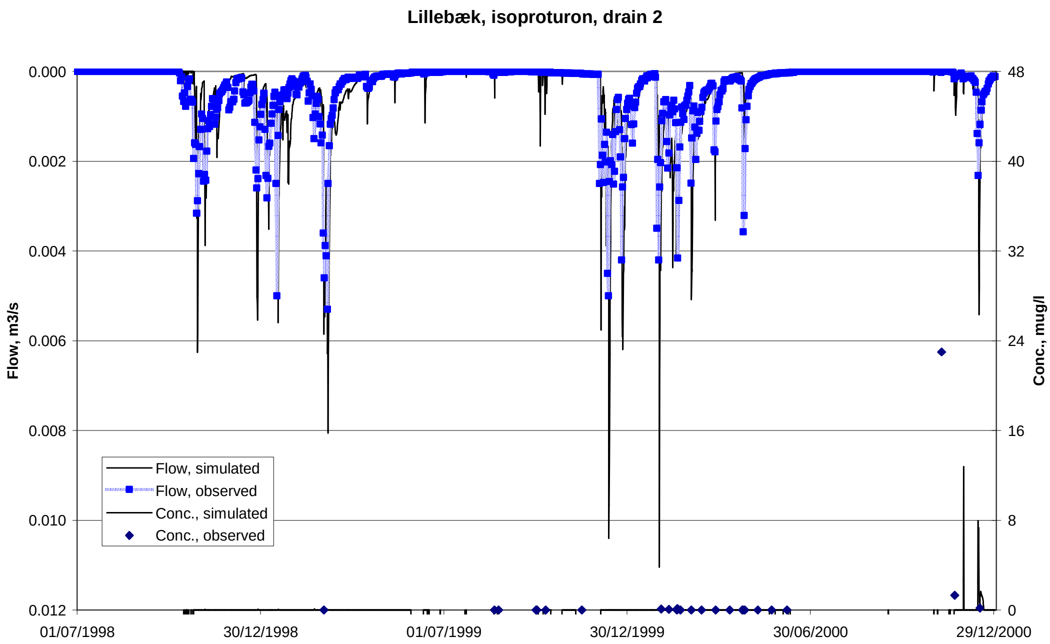
In PestSurf, the Danish EPA requested that all ploughed land be cropped with the same crop and sprayed at the same time. This is an assumption that could have been different. It means that about 85-90 % of the catchments is treated with the same pesticide, and this should, of course, be taken into account when results are evaluated. On the other hand, all natural buffer strips are kept in the simulations. It is possible to define a minimum width of buffer strips in the catchment. Drift and dry deposition occur perpendicular to the stream in both catchments, but only from one side of the stream, of course. The catchments are situated North-Sourth and West –East, and as the prevailing wind is from the west, the two catchments have different susceptibility to drift. However, as the west-east-orientation is not particularly typical, drift was not reduced in the scenarios due to orientation.

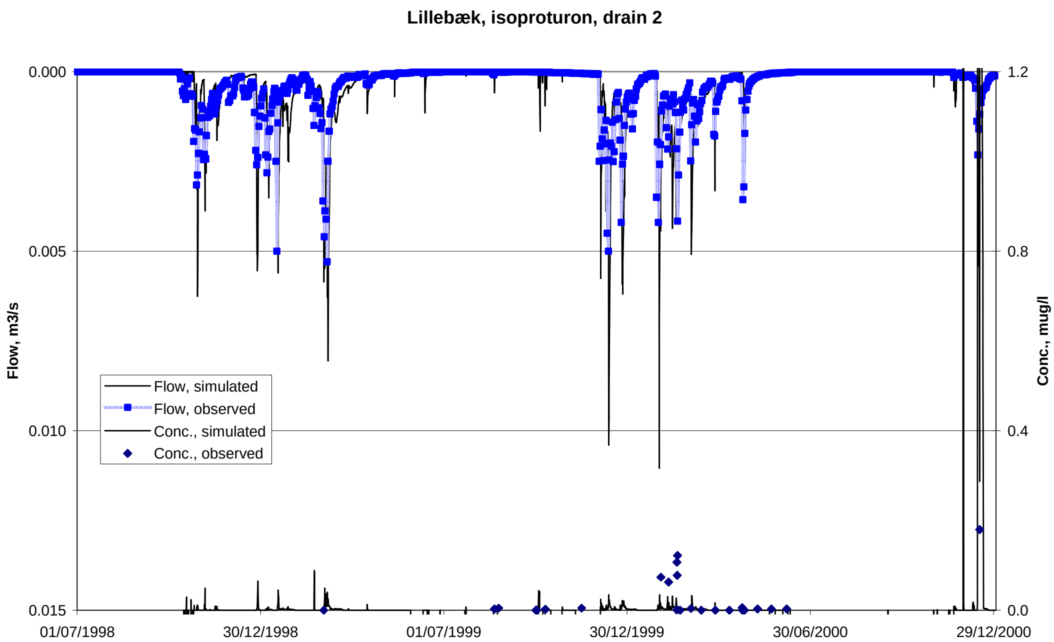
For the calibration exercise, farmers records for the period autumn 1997-mid/end 2000 regarding spraying were used for parameterisation. Pesticide properties stem from the Danish EPA to make the exercise as close to “registration conditions” as possible. No pesticide parameters were measured locally. Four pesticides were found to be used in both catchments, measured in the measurement programme and represented a large span of pesticide properties. These were bentazone, isoproturon, terbuthylazine and pendimethalin.

The results of the calibration exercise did not quite live up to expectations. However, a number of lessons were learned that could be used to improve catchment modelling of pesticides in the future.

First of all, some of the drain simulations were surprisingly good. Particularly the simulation of isoproturon on a drain that received water from a few fields turned out to match observed levels over three years and three orders of magnitude of concentrations (Figure 2.1-6).

Figure 2.1- 6 Simulations and observations of flow and isoproturon at drain 2, shown at two different scales. Concentration levels covered three decades in the period shown



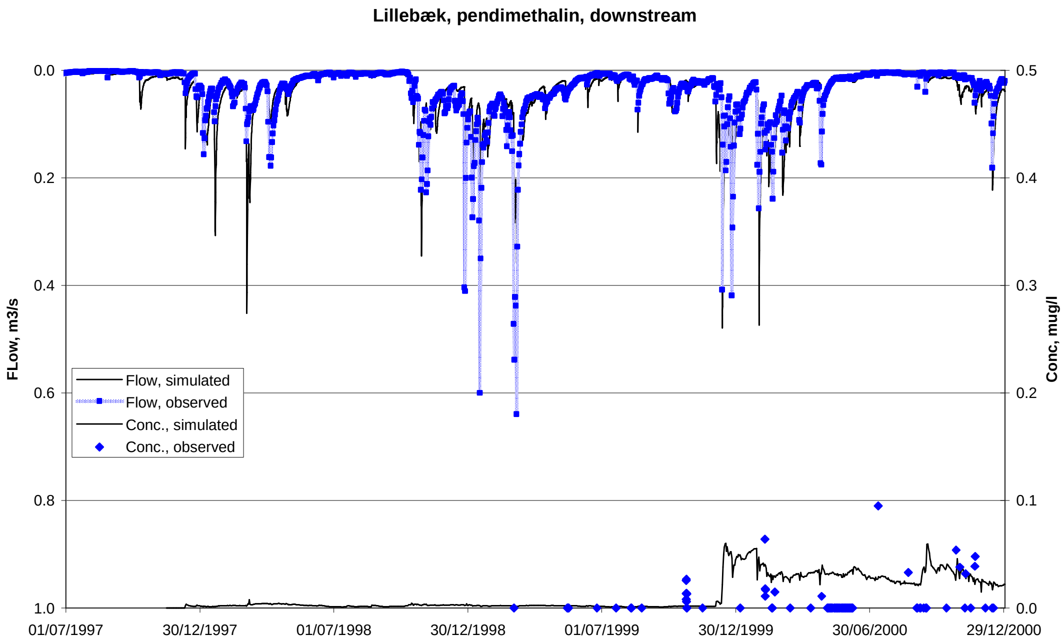


Secondly, the general levels of pesticide simulated were close to but generally lower than the levels observed for the four pesticides (Figure 2.1-7). For the fourth pesticide, terbutylazin, it is thought that the occurrences observed in the stream are due to point sources rather than to the field application. The simulated concentrations are much lower than the observations.

The compound was used on two small fields only upstream of the measuring station but concentrations up to 4.2µg/l were found in the stream.

The shape and the timing of the peaks could, however, be improved. In the sandy catchment, the simulated pesticide levels in the stream showed too little variation. The sandy catchment was chosen to represent an area without macropores. During the study it was found to be more clayey than expected and active macropores were identified. These were not included in the model and the result is clearly seen in the simulations.

Figure 2.1-7 Measured and simulated flow and pendimethalin concentrations in the downstream end of the sandy clay catchment



In the sandy clay catchment, macropores were included and they strongly influenced the simulations. This result is supported by the experimental observations. The colloid transport model included, however, posed strong limitations on how the macropores should be parameterised and it is thought that the present representation of this effect in the model exaggerates the actual extent of solute transport and leads to excessive concentrations moving to groundwater. It must be concluded that at present the understanding of colloidal transport is too limited to allow inclusion at catchment scale.

The timing of the simulated peaks tended to be too late – typically some of the peaks observed during late spring were seen in the simulations only by autumn. Part of the observed fast transport is thus not fast enough in the model. It is thought that in reality, perched water tables are created from time to time and that water moves to the drains from these. In the model, water only drains when the groundwater rises above drain level. This also means that more pesticide is transported to the groundwater in the model than may take place in reality.

Hotspots occurred in the model in areas where high groundwater coincided with particular soil types. The results of the simulation may therefore at times be very dependent on how the distribution of soil types was carried out. However, in general terms, the occurrence of hotspots is likely to be a correct reproduction of reality.

These issues, as well as the issue of discretisation of the upper part of the groundwater producing drainflow points to the fact that a crucial part of the simulation at catchment scale is the boundary between the unsaturated and saturated zone.

When working at catchment scale and calibrating to measured concentrations, one relies on the measured concentrations being reliable. Some of the observed concentrations, however, were judged to be unrealistically high, due to the fact that the pesticides were applied to very small areas only, and concentrations below the root zone would have to be in the mg/l range to create the observed high concentrations. Occurrence of point sources makes it difficult to ensure which observations to trust when comparing with modelling.

<u>Interpretation of catchment model results</u>

A catchment model generates pesticide concentrations in every stream link in the catchment. This means that interpretation of the results becomes more complicated than before. Presently a PEC value for one point is evaluated. Now it may have to be evaluated for many points. This means that it has to be decided not only how much time the critical concentration may be exceeded, but also on how long a stretch. Furthermore, not all stretches may have the same importance due to different recovery potential because some are ephemeral..

Figure 2.1-8 and Figure 2.1-9 show two quite different simulations with PestSurf. About 90% of the catchment area is sprayed with one compound once per year. The first compound builds up in the groundwater and enters the lower part of the stream. As mentioned above, this is a feature that the model is likely to overestimate. The first figure is therefore only included as an example of the complications in the interpretation of data. The upper half of the figure shows the upper 1650 m of the stream. The last four years (the evaluation period) shows two large peaks of which the first occurs in a 20-year-rainfall event, and thus with a long recurrence-period. The second peak occurs in a ”normal” year. In the lower part of the catchment (lowest half of the figure), the concentrations are strongly influenced by an increasing concentration in the groundwater, and the highest concentration is seen in the driest year (last year). The highest concentrations are reached downstream.

In Figure 2.1-8, the highest concentration is reached in the driest year (last year) and in the upper part of the catchment. If recovery is expected to come from upstream, this situation might be considered as more serious. On the other hand, due to the dry conditions, the upper section of the stream has been dry until the day before spraying, and at the time of spraying, the water depth is about 10-15 cm only. This type of situation will happen with a return period of 4-5 years in some parts of the catchment. The question is whether this is a relevant point to evaluate for aquatic risk. The two examples show that the highest concentrations of two different compounds may neither occur during the same year or at the same stretch in the stream, which again complicates the interpretation of results.

It is, however, clear that the more detailed output also necessitates a more sophisticated analysis of risk, including the frequency of occurrence of peaks but also the coverage of a peak in time and space and perhaps also sensitivity and recovery potential at particular sites of a catchment.

Figure 2.1-8. Example of results from a catchment simulation. The upper half of the figure shows the upper 1700 m of the stream, the lower half of the figure, the following 1500 m of the stream.

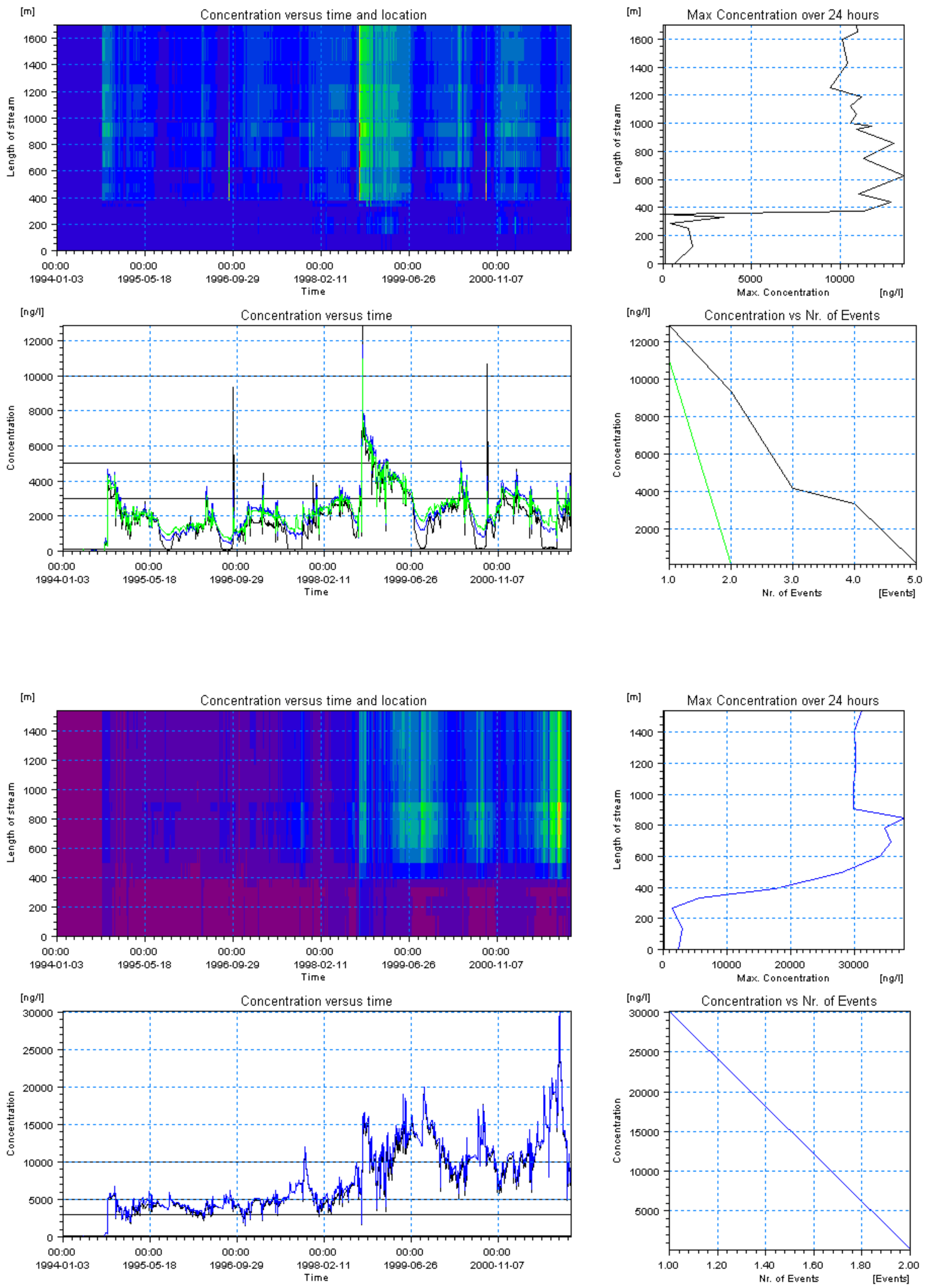
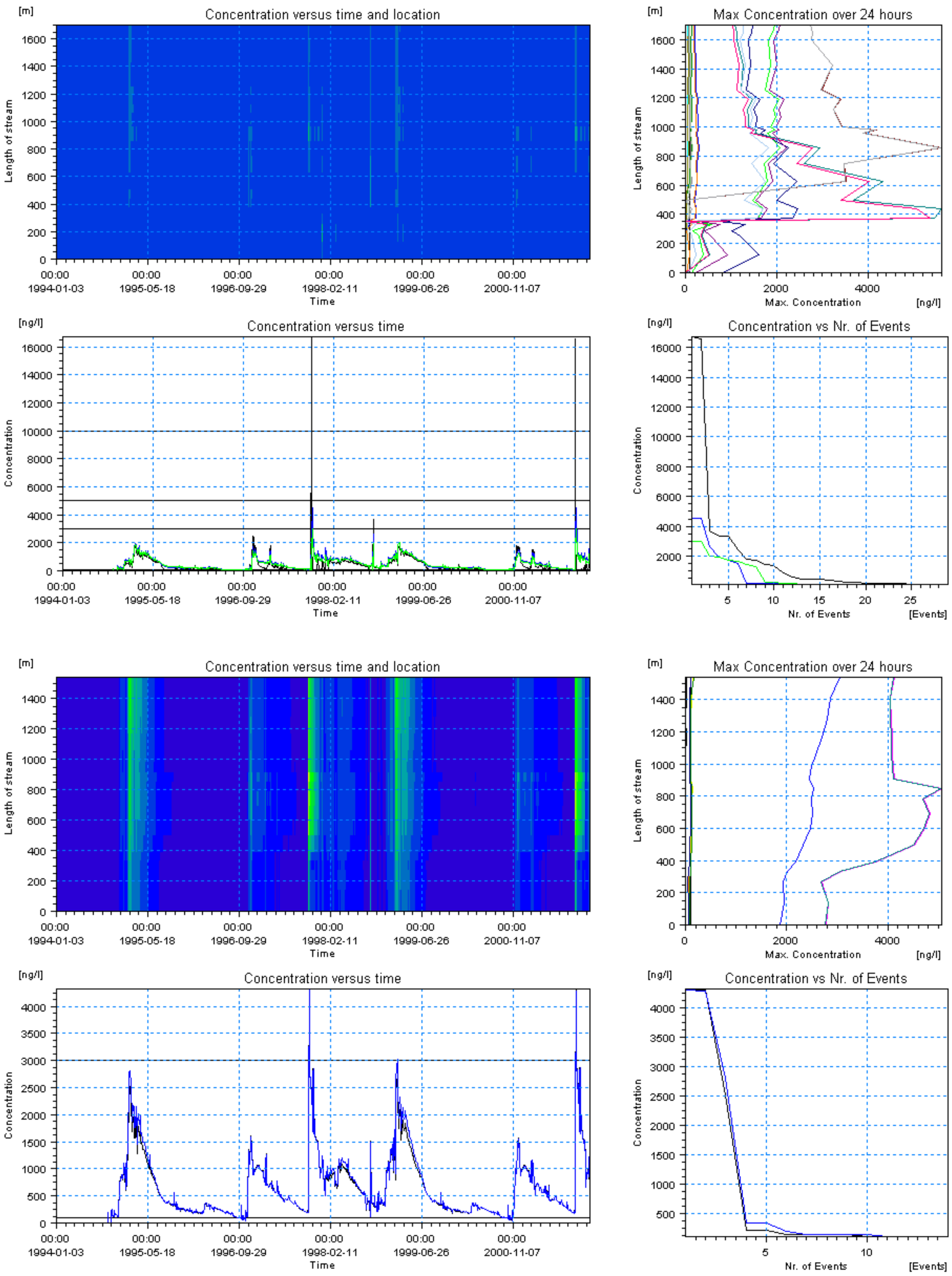


Figure 2.1-9 Example of results from a catchment simulation. The upper half of the figure shows the upper 1700 m of the stream, the lower half of the figure, the following 1500 m of the stream.



Comparisons with the FOCUS surface water scenario-results for selected compounds show that PestSurf generally gives lower concentrations, when the difference in sprayed area is taken into consideration.

<u>Complications of the catchment approach</u>

The drawback of this approach is that it is time consuming to gather the data and calibrate the basic models. It would generally be necessary or at least preferable to select catchments, where monitoring programmes are already ongoing. The two hydrological models created for the Danish catchments have been taken over by the national monitoring programme and will be used also for simulation of the general water balance and nitrate transport and transformation. If monitoring catchments are used as a base for such work, the expenses in data collection, quality assurance of data and modelling may be "shared" between different purposes.

BOX 13

Catchment modelling: Practical Step 4 refinements within the FOCUS modelling framework

Catchment modelling is recommended as a higher-tier assessment tool for use within the FOCUS process. However, due to the complexity of the process and the general lack of readily available models, catchment modelling of aquatic exposures is not likely to find widespread use in the immediate future.

As the capabilities of models and computers are improved, catchment modelling of exposure can be expected to become more routine and it will assume a greater role in regulatory evaluations of pesticides.

2.1.4 Probabilistic modelling

Risk assessment of pesticides is usually performed according to harmonised but deterministic methodologies, where the level of risk is derived from the deterministic quotient of exposure and effects. The parameters to be used for the assessment are selected from a range of values by a pre-set procedure. However, in reality, toxicity and exposure are both distributions of values. Probabilistic methods may be used in order to include the uncertainty of parameter variation in the risk assessment.

Two major European initiatives have resulted in a detailed examination of the various facets of probabilistic modelling of the environmental fate of pesticides: EUPRA and EUFRAM. As a result, this discussion represents a brief overview of probabilistic modelling and the reader is referred to EUPRA and EUFRAM publications for more specific discussions (Hart, 2001 and EUFRAM, 2004).

A typical approach to probabilistic modelling involves multiple model runs with many different sets of parameter values for a number of sensitive parameters. Each parameter is assigned a probability distribution and values for the parameter are sampled accordingly. As the number of parameters increase, the number of combinations increase factorially and methods have been developed to decrease the number of combinations required without adversely impacting the statistical validity of the output.

The advantages of using probabilistic approaches are that (Hart, 2001):

- they help quantify variability and uncertainty,
- they produce outputs with more ecological meaning (e.g. probability and magnitude of effects)
- they make better use of available data
- they assist in identifying the most significant factors,
- they may provide an alternative to field testing or help focus on key uncertainties
- and through the above, the scientific validity is higher.

However, on the other hand, the probabilistic analyses are more complex, they require more data and the results may be difficult to communicate. Furthermore, there is no agreement at present on what outputs are required and how to interpret them. In simple terms, if the exposure level is uncertain and the effect is fixed, the result is a probability distribution of the exposure level and the "risk" is interpreted as the position of the cumulative curve in relation to a vertical line. If both exposure and effects are uncertain, the result is a probability distribution of the quotient "exposure/effects" and risk can be interpreted as the position of the cumulative distribution curve in relation to various ratios of exposure/effects which can be plotted as vertical lines.

Validation of probabilistic approaches is difficult as considerable amounts of monitoring data of very good quality would be required. As with other models, probabilistic approaches only yield correct results if the assumptions on which the analyses are based are correct.

<u>Parameter variability</u>

It must be recognised that there is great variability and uncertainty in field parameters that influence the accuracy of pesticide fate modelling. Spatial variation in pesticide/soil interactions is determined by several factors, many of which remain unexplored. There have been several previous studies on the spatial variation of pesticide/soil interactions (Walker and Brown, 1983; Rao and Wagenet, 1985; Wood *et al.*, 1987; Parkin and Shelton, 1992; Novak *et al.* 1997; Zander *et al*. 1999; Walker *et al*. 2001; Wood *et al.* 2001). These studies focused on quantifying the variation in the sorption (Lennartz 1999) and degradation of pesticides (Walker and Brown 1983; Walker *et al.* 2001; and Wood *et al.* 2001). Walker and Brown (1983) examined spatial variation associated with simazine and metribuzin degradation They showed that small scale variation was an important component of the total variation, by a comparison of the coefficients of variation at different separation distances. Few studies have used geostatistics to quantify the variation in pesticide sorption (Wood *et al.* 1987; Novak *et al*. 1997) or degradation (Parker and Shelton, 1992; Zander *et al.* 1999). This could be due to the large sample size required to compute a reliable variogram. Studies that have applied geostatistics to pesticide/soil interactions have generally been based on small data sets making the computed variograms highly unreliable, e.g. Zander et al., (1999); Parkin and Shelton (1992).

Jury (1986) demonstrated that although there was relatively little variation in bulk density for specific sites (six samples from a specified field would yield a 95% probability in determining a 20% variation in bulk density if it exists (CV = 10%). In contrast, the coefficient of variation (CV) of saturated hydraulic conductivity is significantly higher (CV = 119%) – suggesting that 502 samples would be required from a site to detect a 20% variation in this important parameter for one leaching and run-off models. In studies conducted in

Hawaii by Loague et al. (1990) it was demonstrated that variability in organic carbon content in five soil types was characterised by coefficients of variation in the range of 25-55%. These data demonstrate that geostatistically robust representations of run-off potential are not straightforward and claims regarding representativity should be made with great care.

State of the art in probabilistic modelling

When developing alternative (higher-tier) modelling at Step 4 it may, therefore, become necessary to address some of the more significant uncertainties through a form of probabilistic modelling. One of the more commonly employed techniques that is often used to address the impact of variability is Monte Carlo modelling. PRZM has the capability of carrying out Monte Carlo simulations and limited guidance on the set up of these assessments is provided within the User's manual (Carsel et al., 2003). However, Dubus and Brown (2003) point out that great care should be taken with the design of such assessments. When up to 5000 model runs were undertaken the modelling results were found to be inherently variable (CV of 5 to 211% for 10 replicates). Modelling results were found to be affected by slight changes in the parameterisation of probability density functions and in the assignment of correlation between parameters. Further detailed information on this technique and its regulatory implications is described by Dubus *et al*. (2002) and Warren-Hicks and Moore (1998). These assessments would suggest that with currently available European datasets of key soil parameters there is insufficient background data to address concerns in a statistically robust manner. Nonetheless, probabilistic techniques have great application to assessments of run-off potential and attempts have been made by modellers to develop a robust, systematic approach to avoid some of the more obvious pitfalls of Monte Carlo techniques. As an illustration, the following activities were implemented by the FIFRA Environmental Model Validation Task Force (FEMVTF, 2001) in an effort to ensure the correct implementation of Monte Carlo analysis:

- Strict guidelines were developed for the selection of sampling distributions for the input parameters (see FEMVTF, 2001; Appendix 6)
- Numerous information sources, databases, and experts were identified and consulted in the course of selecting the input parameter sampling distributions,
- A rigorous evaluation of statistical correlation among the input parameters was undertaken

- Comprehensive sensitivity testing of the Monte Carlo outputs was implemented in an effort to ensure results that are not overly dependent upon assumptions and interpretations.

Similarly, the EUFRAM-project, supported by the European Commission’s 5th Framework Programme intends to improve the use of probabilistic approaches for assessing environmental risks of pesticides. The main task of the project (www.eufram.com) is to develop a draft framework on basic guidance for risk assessors, addressing

- the role and outputs of probabilistic assessments
- methods of uncertainty analysis,
- probabilistic methods for small datasets
- methods to report and communicate results
- ways to validate probabilistic methods
- methods to improve access to existing data
- requirements for probabilistic software and databases.

Case studies will be presented, showing how methods can be applied in order to assess impacts of pesticides on terrestrial and aquatic organism. The first draft will be published at the end of 2004.

Probabilistic modelling in the present FOCUS framework

In the present FOCUS framework it is possible to carry out a “manual” probabilistic assessment by running the models multiple times with different sets of parameter values. The selection of parameter values would need to reflect knowledge about the relative likelihood of occurrence of alternative values (e.g. by sampling from a distribution within a Monte Carlo framework). The parameters discussed above make this particularly relevant for the runoff and drainage processes.

With respect to runoff, certain options can be investigated relatively readily within, and outside of, the FOCUS framework with PRZM. Others are more subtle and may require careful justification within alternative scenarios. In order to develop alternative modelling strategies for run-off it is important to initially consider how PRZM simulates run-off. Summaries of PRZM's capabilities are provided by Cohen et al. (1995) and Carsel et al. (2003).

It must be recognised that run-off models may not provide predictions in terms of absolute values to a high degree of accuracy, but can provide very useful information to facilitate relative comparisons between chemicals, application management strategies, site management strategies, rainfall patterns, soils and other variables. It is clear based upon the description above that a wide variety of environmental processes together define the potential for run-off. In order to refine modelling it is important to focus on those environmental input parameters that are likely to have the greatest influence on fate and behaviour. In a modelling exercise considering the sensitivity of PRZM input parameters, Fontaine *et al.* (1992) identified a number of influential parameters as summarised in Table 2.1-6:

Table 2.1-6 Summary of key PRZM input parameters identified by Fontaine(1992)

Important for most ranges
Time between application and rainfall event
Pesticide half-life
K_{oc}
Organic carbon fraction
Available water in surface horizon
Important for many ranges
Runoff curve numbers
Bulk density in surface horizon
Total pesticide applied
Sometimes important
Bulk density in second horizon
Available water in second horizon

With respect to Monte Carlo-simulations of drainage, degradation and sorption parameters will be important, together with the parameters governing the drain flow generation. Dubus and Brown (2002) performed a sensitivity analysis for the MACRO model. As the hydraulic properties are often generated by pedotransfer-functions, it is typically the texture, bulk density and organic content that can be varied. The depth to groundwater and the choice of lower boundary condition is another factor that may influence the results significantly. When macropores are present, the rainfall intensity may be important, as well as the parameters that govern the exchange between the macropores and matrix. As with respect to PRZM, many of these parameters can only be manipulated by going “around” the present FOCUS scenarios, because many of these parameters are fixed.

A broader basis for exposure assessment could be provided by widening the simulation strategy to include a broader range of soils and climate conditions. Further modelling could be conducted under alternative pedoclimatic regimes in a manner similar to that undertaken by Brown *et. al.* (2004). Within this assessment, field monitoring and scenario-based modelling were used to characterise exposure of small ditches in the UK to a herbicide (sulfosulfuron) following transport *via* field drains. A site in central England on a high pH, clay soil was treated with sulfosulfuron and concentrations were monitored in the single drain outfall and in the receiving ditch. MACRO was then used to simulate long-term fate of the herbicide for a broad range of environmental scenarios described below.

The target area (wheat-growing land in England and Wales; *ca.* 1.7×10^6 ha) was divided into environmental scenarios comprising discrete classes of soil type and climate. The soil series making up the drained wheat area were then divided into six broad classes based upon vulnerability for leaching of the acidic herbicide via drainflow. The division was made subjectively based on the relative mobility of sulfosulfuron (determined by soil pH) and the prevalence of rapid movement to drains via macropore flow (determined by clay content and structure). A representative soil series was selected as lying at the vulnerable end of each of the five remaining classes. For each representative series, profile information was extracted from SEISMIC and used to parameterise MACRO.

Areas of wheat cultivation in England and Wales were divided into four approximately equal climatic classes designated 'dry' (<625 mm precipitation per annum), 'medium' (625-750 mm p.a.), 'wet' (750-850 mm p.a.) and 'very wet' (>850 mm p.a.). Four weather datasets were then selected from the SEISMIC database as representative of the four climatic classes. Average annual rainfall for the four datasets was 588, 713, 815 and 1115 mm. The model was then run for the 20 scenarios resulting from the combination of five soil and four climate classes and assuming annual applications of the test compound in the spring of each of 30 years.

Resulting estimates for concentrations of sulfosulfuron in a receiving ditch were weighted according to the prevalence of each scenario to produce a probability distribution of daily exposure.

The described changes to runoff and drainage simulations would to some extent also influence the TOXSWA-simulations. Sorption and dissipation parameters can be manipulated here too. More profound changes of the hydrology would, however, require manipulation of parameters that are presently fixed in the scenarios. The sensitivity of different parameters in stream modelling has been analysed and described in Styczen et al., 2004b, leading to a decision tree for parameter choice based on the actual parameter values.

A description of probabilistic runoff modelling using PRZM has been described by Cohen et al. (1995) and Carsel et al. (2003) involving characterization of both hydrology and transport.

BOX 14

Probabilistic modelling: Practical Step 4 refinements within the FOCUS modelling framework

Probabilistic modelling is recommended as a higher-tier assessment tool for use within the FOCUS process. However, due to the complexity of the process and the general lack of readily available models, probabilistic modelling of aquatic exposures is not likely to find widespread use in the near future except possibly for evaluation of the temporal variation in year-to-year weather.

As the capabilities of models and computers are improved, probabilistic modelling of exposure can be expected to assume an increasingly important role in the regulatory assessment of pesticide exposure.

2.1.5 References

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2.2 Surface water monitoring

2.2.1 Introduction

Monitoring provides real world data on observed exposure and effects following registration and wide scale usage. The objectives of water monitoring vary considerably as pointed out by Trisch and Male (1984):

- To establish a database for planning and development of water resources
- To delineate prevailing water quality conditions and predict possible trends in its quality with respect to time and space
- To provide a basis for the enforcement and development of pollution regulation
- To supply data for the valuation of control and abatement measures
- To provide a database for the development calibration and verification of mathematical models of water quality to be used in support of other activities
- To collect data required for research purposes
- To assure a publicly credible basis for controversial decisions.

In the case of exposure, there are significant surface water monitoring programmes in place throughout Europe designed to pick up prominent pesticides. Monitoring has the capability of being a very useful 'reality check' on exposure predictions but can be difficult to interpret because of wide range of uncertainties such as spatial (what is the proximity to usage area?) and temporal (how long ago was the application or loading?) issues. These uncertainties can be a significant limitation on the value of monitoring databases as a tool within ecological risk assessment.

It is important to note that the size of monitored water bodies will vary depending upon the subsequent use of the collected data. Ecological risk assessments of pesticides are primarily focused on ensuring the safety of aquatic organisms in sites in reasonable proximity to treated fields. As a result, monitoring programmes intended for use in ecological risk assessments of pesticides generally include sampling of small to medium water bodies which drain predominantly agricultural catchments with special emphasis on measuring acute concentrations in the weeks and months immediately following application periods. In contrast, surface water bodies which serve as sources of drinking water generally consist of larger streams, rivers and reservoirs which drain much larger areas. Drinking water monitoring programmes typically involve sampling on a much lower frequency than ecological monitoring studies with a resulting emphasis on establishing chronic exposure levels rather than shorter-term acute concentrations.

2.2.2 Current monitoring programmes in EU and member states

Monitoring data may already be available for the chemical under consideration. Monitoring of residues in drinking water is routinely carried out by Member State environment agencies as well as water authorities, water companies and water service local authorities and various other local government organisations. A number of EU Directives and Decisions have been adopted that require monitoring of rivers for a variety of reasons including (Jowett, 2002):

- Monitoring compliance with environmental quality standards
- Monitoring trends in surface water quality
- Identifying areas susceptible to pollution

The basic legal framework for water quality protection and monitoring is established by EEC Directives 91/271 and 91/676. In the near future the monitoring requirements in all EU Member States will change radically as a result of the implementation of the Water Framework Directive (WFD). Monitoring under WFD will commence at the end of 2006. At present, however, water monitoring throughout Europe is not completely coherent and is established in different Member States in different ways in order to comply with a range of legislative articles.

For example, under the EC Dangerous Substances Directive (76/464/EEC) all regulatory authorities are required to monitor downstream of all known discharges of List I and List II substances. Abstraction points identified under the Surface Water Abstraction Directive (75/440/EEC) are also monitored for ‘relevant’ pesticides. A summary is provided in Table 2.2-1 of the main organisations responsible for collating results of water monitoring programmes in each Member State. A summary is also provided in Tables 2.2-2 – 2.2-16 of water monitoring activities in each Member State. Please note that these Tables were current at the point of initial delivery of this report in spring 2005. This is a rapidly evolving field and there have been developments since this date.

It should be recognised that only a limited number of pesticides are considered within the monitoring programmes described above at any one time. There are significant constraints on monitoring programmes associated with cost, availability, consistency and expense associated with analytical techniques. As a result, monitoring programmes can be very focused, considering specific priority pesticides in key catchments or sub-catchments. For example, water companies in the United Kingdom are required by the Water Supply (Water Quality) Regulations (1989) to carry out monitoring of pesticides in each water supply zone (areas with a population of fewer than 50,000) with the advice that they:

- assess as far as practicable which pesticides are used in significant amounts within the catchment area and;
- assess as far as practicable, on the basis of the properties and method of use of the pesticide and local catchment knowledge, whether any of the pesticides are likely to reach a water source within the catchment area.

In addition to traditional water monitoring programmes, in certain Member States monitoring is also carried out for effects on biota – fishkills and wildlife incidents are recorded and investigated in order to help identify causes and trends. Impact monitoring is very useful in risk assessment as it often highlights unexpected issues not considered in the original evaluation (incorrect usage and disposal etc.).

Table 2.2-1 Summary of organisations responsible for collating results of water monitoring programmes in each Member State

Country	Organization	
Austria	UMWELTBUNDESAMT SPITTELAUER LÄNDE 5 A-1090 WIEN	
Belgium	VMM Vlaamse Milieumaatschappij A. van de Maelestraat 96 B-9320 Erembodegem	DPE Division de la Police de l'Environnement Avenue Prince de Liège 15 B-5100 Namur
Denmark	NERI Ministry of Environment and Energy National Environmental Research Institute P.O. Box 358 Dk-4000 Roskilde	
Finland	National Board of Waters and the Environment Research Institute P.O. Box 250 FIN-00101 Helsinki	

Country	Organization
France	IFEN Institut Francais de l'Environnement 17, rue des Huguenots F-45058 Orléans Cedex 1
Germany	Umweltbundesamt P.O. Box 33 00 22 14191 Berlin
Greece	Ministry of the Environment, Physical Planning and Public Works General Directorate for the Environment Environmental Planning Division Water Section 147 Patission Str. 112 51 Athens
Ireland	Environmental Protection Agency Wexford Ireland
Italy	No information available
Luxembourg	Direction des Eaux et Forets P.O. Box 411 L-2014 Luxembourg Administration de l'Environnement 1a, rue Auguste Lumière L-1950 Luxembourg
The Netherlands	Ministry of Transport, Public Works and Water Management Directorate-General For Public Works Institute for Inland Water Management and Waste Water Treatment, RIZA P.O.box 17 8200 AA Lelystad
Portugal	Ministério do Ambiente e Recursos Naturais Instituto da Água Direccao de Servicos de Recursos Hidricos Avenida Almirante Gago Coutinho, Lisboa
Spain	Ministerio de Obras Públicas, Transportes y Medio Ambiente Secretaria de estado de Medio Ambiente y Vivienda Dirección General de Politica Ambiental
Sweden	Swedish Environmental Protection Agency, Environmental Monitoring and Supervision Department, Monitoring Section, Smidesvägen 5, S-171 85 Solna
United Kingdom	Department of Environment (England and Wales) Environmental Protection Statistics Division Room A104 Romney House 43 Marsham Street London SW1P 3PY Scottish Environmental Protection Agency Erskine Court Castle Business Park Stirling FK9 4TR

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Table 2.2-2 Austrian national surface water monitoring programmes (Kristensen and Bøgestrand, 1996)

No.	Name	Responsible institution	Variables	Period of operation & Sampling Frequency (GREENED)	Geographical coverage	Data & reporting
Rivers and streams						
R1	Ordinance on Water Quality Monitoring	MAF	physical, chemical, bacteriological & biological variables	since 1991 SF: 6/yr	244 sampling sites at national rivers	Database & reporting; MAF-WMR & FEA
R2-R8	Water quality monitoring of transboundary rivers 2. Bucharest Declaration 3. Regenburger Vertrag 4. AU-CZ Grenzgewässer-kommision (GK) 5. AU-SK GK 6. AU-HU GK 7. AU-SL GK river Mur 8. AU-SL GK river Drau	BfW & the commissions, respectively	physical, chemical & biological variables	2. since 1988 SF: monthly 3. AU-DE since 1991 SF: monthly 4. AU-CZ since 1968 SF: 1-4/yr 5. AU-SK since 1968 SF: every 2 months 6. AU-HU since 1972 SF: 2-12/yr 7. AU-SL since 1965 SF: 2/yr 8. AU-SL since 1955 SF: 1/yr	2. 2 sites at Danube 3. Rivers crossing AU-DE border 4. Rivers crossing AU-CZ border 5. Rivers crossing AU-SK border 6. Rivers crossing AU-HU border & Lake Neusiedler See 7. River Mur 8. River Drau	AU database; BfW Database & reporting by the commissions, respectively
Lakes and reservoirs						
L1	Water quality monitoring according to the "transboundary commission" for Lake Constance	IKGB	physical, chemical & biological variables	since ? SF: differs according to the special monitoring programme	Lake Constance & tributaries	Database & reporting; IKGB
L2	Gewässergüteuntersuchungen Zeller See	BfW	physical, chemical, bacteriological & biological variables	since 1953 SF: 5/yr	Lake Zeller See	Database & reporting; BfW

AU: Austria; SK: Slovak Republic; SL: Slovenia; DE: Germany; CZ: Czech Republic; HU: Hungary
MAF: Federal Ministry of Agriculture and Forestry; WMR: Water Management Register; FEA: Federal Environmental Agency;
BfW: Bundesanstalt für Wassergüte; IKGB: Internationalen Gewässerschutz-Kommission für den Bodensee

Table 2.2-3 Belgian surface water monitoring programmes (Kristensen and Bøgestrand, 1996)

No.	Name	Responsible institution	Variables	Period of operation & Sampling Frequency (SF)	Geographical coverage	Data & national reporting
Rivers and streams						
R1	Physico-chemical monitoring network	VMM	11-16 physical and chemical variables	Since 1989 SF: 8-12/yr, sometimes 26-52/yr	Flanders region 1000 sampling sites	Database and annual report: VMM
R2	Biological monitoring network	VMM	Macroinvertebrates, Belgian Biotic Index	Since 1989 SF: 1/yr.	Flanders region 900 sampling sites	Database and annual report: VMM
R3	Measuring network suspended solids	VMM	Water & suspended solids: organic micropollutants & heavy metals	Since: Sep. 1994 SF 3/yr	Flanders Region 20 sampling sites	Reports after each campaign
R4	Measuring network water soils	VMM	Heavy metals & organic micropollutants	Study undertaken in 1991-92 and will be repeated in 1995-1996	Flanders Region	Reports after each campaign
R5	Physico-chemical monitoring of surface waters	DPE	Up to 108 physical and chemical variables	1. network: 5/yr. 2. network: 12/yr	Walloon region 1.network: 90 sites covering main rivers, streams, canal, and reservoirs. 2. network 7 sites on transbordering water courses	Database & annual report: DPE.
R6	Physico-chemical monitoring of designated protected waters 3 networks 1) Freshwater for fish 2) Surface water for drinking water 3) Natural water networks	DPE	16-28 physical & chemical variables	1) 12/yr 2) 2-8/yr 3) 12/yr	Walloon region. 1) 38 sampling sites, 2) 5 sampling sites 3) 6 sampling sites	Database & annual report: DPE.
R7	Alarm network.	DPE	6-12 physical & chemical variables	continuous (every 3 minutes)	River Meuse, 3 sites. River Sambre, 2 sites.	Database & annual report: DPE.

No.	Name	Responsible institution	Variables	Period of operation & Sampling Frequency (SF)	Geographical coverage	Data & national reporting
R8	Biological quality assessment of water courses	1980-92:IHE 1993-: DPE	Macroinvertebrates, Belgian Biotic Index, 5 physical & chemical variables	once every 3 years. case-studies: according to occurrence	Walloon region. 200+150+60 sampling sites	Database & annual report: DPE.
R9	Bathing water	1982-89: IHE 1990-: DPE	physical, chemical & microbiological variables	9 samples from mid-May to mid-September	Walloon region 47 sampling sites in fresh surface water	Database & annual report: DPE.
R10	Hydrometric networks.	DPE & SETHY	flow, water height	4 measures per hour	Walloon region 103+145 hydrometric stations.	Database: DPE & SETHY

Table 2.2-4 Danish national surface water monitoring programmes (Kristensen and Bøgestrand, 1996)

No.	Name	Responsible institution	Variables	Period of operation & Sampling Frequency (SF)	Geographical coverage	Data & national reporting
Rivers and streams						
R1	Nation-wide Monitoring Programme Monitoring of streams	NERI	Chemical and physical variables Macroinvertebrates	Since 1989 SF: 12-26 (52)/yr	Nation-wide, 261 sampling sites in approx. 125 river systems	Database: NERI Reporting: NERI
R2	Nation-wide Monitoring Programme Monitoring of springs.	NERI	Chemical and physical variables	Since 1989 SF: 4/yr	Nation-wide 58 springs	Database: NERI Reporting: NERI
R3	Nation-wide Monitoring Programme Monitoring of agricultural watersheds	NERI	Chemical and physical variables on soil water, drainage water, ground water and river water.	Since 1989 SF: 12-26 (52)/yr	6 agricultural watersheds	Database: NERI & GSD Reporting: NERI & GSD
R4	Inventory of biological assessment of river quality	EPA	Macroinvertebrates Quality classification grades	Since 1989 SF: 1-2/yr	Nation-wide. 10,000 sampling sites	Database and reporting: EPA
Lakes						
L1	Nation-wide Monitoring Programme Monitoring of lakes	NERI	Chemical and physical variables in lake water and tributaries. Phyto- & zooplankton, fish and macrophytes. Sediment composition	Since 1989 SF: Lake water 19/yr Tributaries 12-26/yr Plankton 19/yr Fish, macrophytes & sediment 1/5 yr	Nation-wide 37 lakes	Database: NERI Reporting: NERI
Coastal and marine areas						
M1	Nation-wide Monitoring Programme Monitoring of coastal and open marine waters	NERI	Chemical and physical variables. Phyto- & zooplankton, zoobenthos and macrophytes. Sediment composition	Since 1989 SF: Water 8-52/yr Plankton 8-52/yr Zoobenthos 1/yr Macrophytes & sediment 1/5/yr	Nation-wide 200 coastal sampling sites and 80 offshore sampling sites.	Database: NERI Reporting: NERI

NERI: National Environmental Research Institute, Ministry of Environment and Energy, GSD: Geological Survey of Denmark, Ministry of Environment and Energy

Table 2.2-5 Finnish national surface water monitoring programmes (Kristensen and Bøgestrand, 1996)

No.	Name	Responsible institution	Variables	Period of operation & Sampling Frequency (SF)	Geographical coverage	Data & reporting
Rivers and streams						
R1	Water quality at river streamflow stations.	NBWE & WERI	41 physical & chemical variables	Since 1962: SF: 4/yr between March-October	National 68 sampling sites	Database; NBWE. 1 report/yr; WERI
R2	Transport of suspended & soluble material from land areas	NBWE & WERI	18-26 physical & chemical variables	Since 1962: SF: 1/wk in spring, 2/mon. in autumn	15 small drainage basins	Database; NBWE. 1 report/yr; WERI
R3	Material input to the Baltic Sea by Finnish rivers	NBWE & WERI	41 physical & chemical variables	since 1970: SF: min. 12/yr	seacoast: 30 stations Rivers: average flow > $5 m^3 s^{-1}$	Database; NBWE 1 report/yr; WERI
R4	Monitoring of water quality in the bordering rivers of Finland	The trans-boundary water commissions, NBWR & WEDs	physical & chemical variables	FI-RU; since 1964, SF; 4-12/yr FI-NO; since 1980 SF; 7/yr FI-SE; since 1976; SF; 12/yr	FI-RU: 8 sites FI-NO: 1 site FI-SE: 3 sites	Report of information; WERI
Lakes and reservoirs						
L1	Water quality in lake deeps	NBWE & WERI	28 physical & chemical variables	since 1962: SF: 3/yr	National 71 sampling sites	Database; NBWE 1 report/yr; WERI
L2	Biological monitoring of inland waters	NBWE & WERI	biological variables	since 1963: SF: every 3'rd yr	National 71 sampling sites	Database; NBWE 1 report/yr; WERI
L3	Monitoring of bioaccumulating compounds in fresh waters & environ'l specimen bank	NBWE & WERI	heavy metals, organic compounds, pesticides	since 1978: SF: every 2'nd or 3'rd yr	major rivers & lakes	Database; NBWE 1 report/yr; WERI
L4	Acidification monitoring of surface waters	NBWE & WERI	25 physical & chemical variables	since 1987: SF: 1/yr	National, 176 + 200 lakes	Database; NBWE 1 report/yr; WERI
Coastal and marine areas						
M1	Monitoring of coastal Finnish waters	WERI, Research Laboratory & WEDs	24 physical & chemical variables, biological variables, heavy metals, organic compounds, pesticides	since 1964, 1966, 1978 depending on the parameter: SF 1-20/yr depending on the parameter	12 intensive stations, 94 other stations	Database; WERI, FIMR. Report; every 5 yr; HELCOM
M2	Monitoring of open sea waters	FIMR & GFRI	24 physical & chemical variables, biological variables, heavy metals, pesticides	since 1979: SF; daily to 4/yr depending on parameter	All main deep basins in Gulf of Bothnia, Gulf of Finland & the Baltic proper	Database; FIMR & NBWE Report every 5 yr

FI: Finland; NO: Norway; SE: Sweden; RU: the Russian Federation. NBWE: National Board of Waters and the Environment; WERI: Water and Environment Research Institute; FIMR: Finnish Institute of Marine Research; GFRI: Game and Fisheries Research Institute; WEDs: 13 Water and Environment Districts; HELCOM; Helsinki Commission

Table 2.2-6 French national surface water monitoring programmes (Kristensen and Bøgestrand, 1996)

No.	Name	Responsible institution	Variables	Period of operation & Sampling Frequency (SF)	Geographical coverage	Data & reporting
Rivers and streams						
R1	Inventory of the quality of running fresh waters	RNB	47 physical & chemical variables	since 1987 SF: min. 8/yr	National 1082 sampling sites	Database; RNB Report; RNDE, RNB
Lakes and reservoirs						
	No national network exists					No national data storage
Coastal and marine areas						
M1	National sea water quality monitoring network - RNO	RNO	basic components, enzymatic activity, metals & pesticides	since ? SF: water; 2-12/yr biomass; 4/yr sediment; every 2-5 yr	43 areas, each composed of several sampling sites	Database; RNO annual reports; IFREMER
M2	French seashore microbiological monitoring – REMI	REMI	faecal coliform, salmonellas		314 sampling sites in 88 areas	database; REMI/REPHY Report; IFREMER
M3	French seashore phytoplankton monitoring – REPHY	REPHY	phytoplankton species composition, toxicity	SF: twice a month, alert monitoring on a weekly basis	37 sampling sites; alert programme 70-80 sites	database; REMI/REPHY Report; IFREMER

RNB: National Basin Network; RNO: National sea water quality monitoring network; IFREMER: Institut Francais de Recherche pour l'Exploitation de la Mer; REMI: French seashore microbiological monitoring; REPHY: French seashore phytoplankton monitoring. network.

Table 2.2-7 German national surface water monitoring programmes (Kristensen and Bøgestrand, 1996)

No.	Name	Responsible institution	Variables	Period of operation & Sampling Frequency (SF)	Geographical coverage	Data & reporting
Rivers and streams						
R1	Überwachungsprogramm der Länderarbeitsgemeinschaft Wasser (LAWA)	FS, LAWA (UB for data collection)	Physical & chemical variables.	Since 1982 SF: 13/yr	Nation-wide 146 sampling sites	Database: QUADAWA(UB) Report every 5 yrs:LAWA
R2	Biological classification of the quality of inland surface waters (rivers)	LAWA	Biological: INVERT Saprob.index	Since 1975 SF: once within 5 yrs	Nation-wide	Database: no national database Report every 5 yrs:LAWA
Coastal and marine areas						
M1	Bund/Länder-Messprogramm für die Nordsee	ARGE	Physical, chemical and biological variables	Since 1980 SF: 1-4/yr	53 sampling sites in the North Sea	Database: MUDAB Report: ARGE
M2	Bund/Länder-Messprogramm für die Ostsee	ARGE	Physical, chemical and biological variables	SF: 5-11/yr	Baltic Sea and Baltic Proper	Database: MUDAB Report: IOW

FS: Federal States (Länder); LAWA: Joint Water Commission of the Federal States ('Länderarbeitsgemeinschaft Wasser'); UB: Umweltbundesamt; ARGE: Bund/Länder-Messprogram Committee, Federal Ministry of the Environment; MUDAB: Marine data base (Meeresumwelt Datenbank); QUADAWA: River Water Database ('Qualitätsdatenbank Wasser'); IOW: Institut für Ostseeforschung, Warnemünde.

Table 2.2-8 Greek national surface water monitoring programmes (Kristensen and Bøgestrand, 1996)

No.	Name	Responsible institution	Variables	Period of operation & Sampling Frequency (SF)	Geographical coverage	Data & reporting
Inland surface waters						
R1	Monitoring of surface water quality	L.S.G.	organic & inorganic chemical variables physical variables	since early 1980s SF: monthly & seasonally	Greek surface waters	Database; L.S.G. annual reports, WS
R2	National Monitoring Programme for Surface Waters	G.C.S.L.	physico-chemical variables	not in operation yet	surface waters of the corresponding district	Database; G.C.S.L. annual reports, WS
Coastal and marine areas						
M1	MED POL in the Aegean and Ionian Sea MED POL in the Saronic Gulf	N.C.M.R.	organic & inorganic chemical variables, biological & physical variables	since 1985: SF: seasonally, 4/yr	Saronic Gulf, Aegean & Ionian Sea	Database; N.C.M.R. biannual reports; WS
M2	MED POL, Cretian marine waters	I.M.B.C.	organic & inorganic chemical variables	since 1988: SF:seasonally, 3/yr	Cretian marine waters	Database; I.M.B.C. biannual reports;WS
M3	Bathing waters Determination of possible pollution problems	WS	biological variables	since 1988: SF: fortnightly (May-October)	Greek bathing areas	Database & annual reports; WS

WS: Water Section, Ministry of Environment; L.S.G.: Laboratory of Soil-hydrology and Geology, Ministry of Agriculture;

G.C.S.L.: General Chemical State Laboratory; I.M.B.C.: Institute of Marine Biology of Crete; N.C.M.R.: National Centre for Marine Research

Table 2.2-9 Irish national surface water monitoring programmes (Kristensen and Bøgestrand, 1996)

No.	Name	Responsible institution	Variables	Period of operation & Sampling Frequency (SF)	Geographical coverage	Data & national reporting
Rivers and streams						
R1	Physico-chemical Surveys of River Water Quality	EPA Local Authorities	Water: Physical and chemical variables at some sampling sites measurements of metals	Since 1970-71 SF: varying (12/yr)	Nation-wide. Mainly large rivers and primary tributaries	Database: EPA & local authorities. Reporting: every three years by EPA
R2	National Biological Survey of River Water Quality.	EPA	Water: TEMPW, OX Biological: INVERT, MAPHYT, Filamentous algae, Siltation	Since 1971 Every 3rd year or more frequently	Nation-wide. 3000 sampling sites in 1200 rivers and streams	Database: EPA Reporting: Every 3rd year by EPA
R3	The Recording of Fish Kills	DoM	Fish kills and if possible, their causes	1971-1974 and 1983 to date	Nation-wide. No specific network	Reporting annually by DoM
Lakes						
L1	Lake Water Quality Monitoring Programme. (a)In situ measurements. (b)Remote sensing surveys	EPA & Local Authorities, RPII, CFB	(a) Water: Chemical & physical variables (b) Remote sensing	(a)Since the late 1960s SF: from several times per year to 1/3-5yr (b) 1989-1990	(a) Nation-wide. Large lakes and representative smaller lakes. 170 lakes. (b) Nation-wide. 360 lakes	Data and reporting: EPA & local authorities. Every three years
Coastal and marine areas						
M1	General Quality of Estuarine and Coastal Receiving Waters Including Nutrients.	FRC, EPA, DoM & Local authorities	Water: Physical and chemical variables	Since 1992. 1 winter survey and a number of surveys in summer	Nation-wide. Significant estuaries & coastal areas and the Western Irish Sea	Reporting: 1/4 yr by EPA, DoM & local Authorities
M2	Metals and organic micropollutants in the Estuarine and Coastal Environment.	EPA, FRC/DoM (MI), Local Authorities	Water: organic micropollutants Sediment & biota: heavy metals & organic micropollutants	Since 1993 One major estuary per year in a 5-6 year cycle. Trend monitoring of metals in mussels	Nation-wide	Reporting by FRC to the JMG

No.	Name	Responsible institution	Variables	Period of operation & Sampling Frequency (SF)	Geographical coverage	Data & national reporting
M3	Radioactivity Monitoring of the Irish Marine Environment.	RPII	Radionucleides in water, sediment, & biota	Since the early 1970s. SF: 2-4/yr.	Nation-wide. Greatest density of sites where the impact of the Sellafield facility is greatest.	Reporting: 1/2yr by RPII
M4	Environmental Quality of Amenity and Recreation Areas, in particular, Bathing Waters	DoE Local Authorities	Water: Physical, chemical, & microbiological variables	Since 1979 SF: 1/1-2 week from mid-May to ultimo August	Nation-wide. A total of 92 important marine bathing areas	National reporting annually by DoE
M5	Bacteriological Quality of Shellfish Waters.	DoM	Faecal coli in water and shellfish.	Since 1981 SF: 2 weeks intervals throughout the year	Mainly W and SW coast. 200 locations in 50 coastal inlets	DoM
M6	Monitoring of Human Food Sources.	DoM/MI, (FRC)	Water: Physical variables Shellfish: metals & organic micropollutants Fish: HG	Since 1992 SF: Annually	Nation-wide. 18 shellfish growing waters and 5 important fishing ports	FRC, JMG

EPA: Environmental Protection Agency; DoM: Department of the Marine; RPII: Radiological Protection Institute of Ireland; MI: Marine Institute; FRC: Fisheries Research Centre; JMG: Joint Monitoring Group; DoE: Department of the Environment; CFB: Central Fisheries Board.

Table 2.2-10 National surface water monitoring programmes of Luxembourg (Kristensen and Bøgestrand, 1996)

No.	Name	Responsible institution	Variables	Period of operation & Sampling Frequency (SF)	Geographical coverage	Data & reporting
Rivers and streams						
R1	Biochemical monitoring programme of rivers	AE	26 chemical and physical variables	SF: 1-13/yr	Nation-wide 217 sampling in all main rivers	
R2	Biological management and control of inland waters	AEF, SCP	PHYTPL, ZOOPL, INVERT, MAPHYT, FISH	Since 1972 Heavily polluted: 1/year Others: every 3-5 years	Main rivers and principal affluents of the whole country	Reporting: SCP
Lakes						
L1	National Lake Monitoring Programme		22 chemical, physical and microbiological variables	SF: 8/yr	10 sampling sites in 3 lakes	

AE: Administration de l'Environnement; AEF: Administration des Eaux et Forêts; SCP: Service Chasse et Pêche

Table 2.2-11 Dutch national surface water monitoring programmes (Kristensen and Bøgestrand, 1996)

No.	Name	Responsible institution	Variables	Period of operation & Sampling Frequency (SF)	Geographical coverage	Data & reporting
Inland surface waters						
R1	National Surface Water Monitoring Programme (MWTL) Monitoring of Inland Waters	RIZA	120 chemical, physical and biological variables	Since 1955 SF: Chemical & physical variables 6-52/yr, biological variables 1-13/yr	Presently 26 sites throughout the country	Data storage and yearly reporting by RIZA
R2	Aqualarm Early warning network	RIZA	Chemical & physical variables	Since 1974 (semi-) continuous	7 online stations along the rivers Rhine & Meuse	No reporting
Coastal and marine areas						
M1	National Surface Water Monitoring Programme (MWTL) Monitoring of Marine Waters	RIKZ	Chemical, physical and biological variables	Since 1972 SF: chemical & physical variables 1-13/yr, biological variables 1-18/yr	95 sites along the coast	Data storage and yearly reporting by RIKZ

RIZA: Institute for Inland Water Management and Waste Water Treatment; RIKZ: Institute for Coast and Sea

Table 2.2-12 National surface water monitoring programmes in Portugal (Kristensen and Bøgestrand, 1996)

No.	Name	Responsible institution	Variables	Period of operation & Sampling Frequency (SF)	Geographical coverage	Data & national reporting
Rivers and streams						
R1	Water Quality Network Rede de qualidade da água	INAG & DRARN	Chemical and physical variables	Since SF: monthly	Nation-wide, 109 sampling sites in primarily large rivers	Database: DRARN Reporting: INAG

INAG: National Institute of Water; DRARN: Regional Directorate of Environment and Natural Resources

Table 2.2-13 Spanish national surface water monitoring programmes (Kristensen and Bøgestrand, 1996)

No.	Name	Responsible institution	Variables	Period of operation & Sampling Frequency SF	Geographical coverage	Data & national reporting
Rivers and streams						
ES-R1	Assessment of physico-chemical river quality	MOPTMA	Chemical and physical variables	Since 1962 SF: 1-12/yr	Nation-wide 448 sampling sites in all main Spanish rivers	Database: MOPTMA Reporting annually
ES-R2	Biological classification of river water quality	MOPTMA (CEDEX)	Macroinvertebrates	Since 1980 SF: 4/yr	Nation-wide 847 sampling sites in all main Spanish rivers. 160 sites/yr	Database: MOPTMA Reporting annually
Lakes and reservoirs						
ES-L1	National survey on eutrophication in reservoirs	MOPTMA (CEDEX)	Physical, chemical and biological variables	Since 1972 SF: 4/yr	Nation-wide 350 reservoirs	Database: MOPTMA Reporting annually
ES-L2	National survey on eutrophication in reservoirs by remote sensing	MOPTMA (CEDEX)	Temperature, transparency, chlorophyll	Since 1984 SF: 4/yr in summer	Nation-wide 496 reservoirs, 1 river basin each year	Database: MOPTMA Reporting annually

MOPTMA: Ministerio de Obras Públicas e Urbanismo, Dirección General de Obras Hidráulicas; CEDEX: Centro de Estudios y Experimentaciónde Obras Públicas del MOPTMA.

Table 2.2-14 National surface water monitoring programmes in Sweden (Kristensen and Bøgestrand, 1996)

No.	Name	Responsible institution	Variables	Period of operation & Sampling Frequency	Geographical coverage	Data & reporting
Lakes and streams						
L1/R1	National - lake & Stream survey	SUAS	physical & chemical variables, macroinv.	since 1972; SF: every five yr	National - 1000 lakes, 300 streams	database; SUAS Report every fifth yr; SUAS
L2/R2	National time-series in reference lakes & streams	SUAS	physical, chemical & biological variables, sediment, palaeoreconstruction	since 1960s SF: 4-12/yr, depends on parameter	National - 85 lakes, 35 streams	database; SUAS Report 1/yr; SUAS
L3/R3	National intensive time-series in reference lakes & -streams	SUAS	physical, chemical & biological variables, sediment, paleoreconstruction, contaminants in fish	since 1960s SF: 1-12/yr, depends on parameter	National - 15 lakes, 15 streams	database; SUAS Report 1/yr; SUAS
Rivers						
R4	National main-river outlets	SUAS	physical & chemical variables	since 1960s SF: monthly	49 main rivers	database; SUAS Report 1/yr; SUAS
Coastal and marine areas						
M1	National pelagic high frequency monitoring	UMSC, SMSC, GMSC	physical, chemical & biological variables	GB: since 1989 BP: since 1976 K&S: since 1993 SF: 8-25/yr	three coastal & five offshore stations	Database & annual report; UMSC, SMSC, GMSC, SMHI & RSAS
M2	National pelagic frequent monitoring	UMSC, SMHI	physical & chemical variables	since 1992; GB since 1993; BP, K&S SF: 6-12/yr	GB; 10 stations BP; 12 stations K&S; 3-4 stations	Database & annual report; UMSC & SMHI
M3	National pelagic low frequent monitoring	SMHI	physical, chemical & biological variables	since 1993 SF: 6-12/yr FS SF: 1/yr LFS	GB, BP, the Sound, K&S; FS: 25 stations, LFS: 68 stations	Database & annual report; SMHI
M4	National soft bottom macrofauna	UMSC, SMSC, GMSC	zooben, splist, cnr, EH, sal, tempw, macrofauna, sediment	since: BP; 1980, GB, K&S; 1983, SF: 1/yr May-June	offshore & coastal waters	Database & annual report; UMSC, SMSC, GMSC
M5	National phytobenthos	SMSC, GMSC	plants & animals, substratum, sal, secchi.	since ? SF: 1/yr August	The Baltic Proper & the Skagerrak	Database & annual report; SMSC, GMSC

No.	Name	Responsible institution	Variables	Period of operation & Sampling Frequency	Geographical coverage	Data & reporting
M6	National malformed embryos of Monoporeia affinis	IAER	no. of eggs & abnormal embryos	since 1993 SF: 1/yr February	5 stations in GB, 7 stations in nBP	Database & annual report; IAER
M7	National ecological coastal fish monitoring	NBFI	fish stock & individual analysis	since 1989 SF: 1-2/yr	one coastal area in BP, GB & the Skagerrak	Database & annual report; NBFI
M8	National physiological coastal fish monitoring	IAER, GMSC	blood & tissue constituents	since 1989 SF 1/yr summer	one coastal area in BP, GB & the Skagerrak	Database & annual report; IAER, GMSC
M9	Contaminant monitoring programme	IAER, SMNH, SUAS	contaminants (heavy metals, pesticides) in biota	since 1979 SF 1/yr in autumn	GB, BP, K&S	Database & annual report: IAER, SMNH, SUAS
M 10	Monitoring of top predators (seals and eagles)	SMNH	Population size and dynamics. Health status.	Since 1989 SF 1/yr	GB, BP, K&S	Database & annual report: SMNH

GB: Gulf of Bothnia; nBP: northern Baltic Proper; BP: Baltic Proper; K&S: Kattegat & the Skagerrak; FS: Frequent Sampling; LFS: Low Frequent Sampling; SUAS: The Swedish University of Agricultural Science, Department of Environmental Assessments; UMSC: Umeå Marine Science Centre; SMSC: Stockholm Marine Science Centre; GMSC: Gothenburg Marine Science Centre; IAER: Institute of Applied Environmental Research, University of Stockholm; SMHI: The Swedish Meteorological & Hydrological Institute; NBFI: The National Board of Fisheries, Institute of Coastal Research; SMNH: The Swedish Museum of Natural History, Contaminant Research Group.

Table 2.2-15 National surface water monitoring programmes in the United Kingdom (Kristensen and Bøgestrand, 1996)

No.	Name	Responsible institution	Variables	Period of operation & Sampling Frequency (SF)	Geographical coverage	Data & reporting
Rivers and streams						
R1	The Harmonised Monitoring Programme	DoE, NRA SOEnD, RPB	over 80 physical and chemical attributes of river quality, but typically only 25 are measured at any given site	Many sites since 1975 SF: 6-52/yr	A national network covering Great Britain 220-230 sampling sites	
R2	General Quality Assessment (GQA) Chemical assessment of rivers, canals and lochs	NRA SOEnD, RPB DoE(NI)	OX, OXSAT, BOD5, NH4N, and variables appropriate to the stretch in question	Since 1976 SF: 12/yr (4-24/yr)	England & Wales: 40,000 km of rivers and canals, approx. 7,000 sites Scotland: 50,000 km of rivers and canals, approx. 2800 sites Northern Ireland: 2,500 km of rivers, approx. 290 sites	Database: NRA, SOEnD, DoE(NI) Reporting: NRA, SOEnD, DoE(NI)
R3	General Quality Assessment (GQA) Biological classification of rivers	NRA SOEnD, RPB DoE(NI)	macroinvertebrates	Start year: 1990 Every 5 year two or three annual samples	England & Wales: 40,000 km of rivers and canals, approx. 7,000 sites Scotland: 11,000 km of rivers, 976 sampling sites Northern Ireland: 2,500 km of rivers, approx. 290 sites	Database: NRA, SOEnD, DoE(NI) Reporting: NRA, SOEnD, DoE(NI)
Lakes and reservoirs						

No.	Name	Responsible institution	Variables	Period of operation & Sampling Frequency (SF)	Geographical coverage	Data & reporting
L1	Blue-green Algae Annual Sampling Programme	NRA DoE(NI)	Blue-green algae, water samples, bloom and/or scum material	England & Wales: Start year 1989 Routine sampling and reactive sampling Northern Ireland: 1993: Routine monitoring programme From 1994: only reactive monitoring Scotland: Routine monitoring	England & Wales: NRA regions Northern Ireland: 1993: 52 water abstractions and 17 recreational waters. Scotland: Waters considered to be at risk.	Data held by NRA & DoE(NI), no public report
L2	Monitoring of inland waters commonly used for recreation	DoE(NI)	Microbiological indicators & blue-green algae	Since 1992 SF: 5/yr	Northern Ireland 14 waterbodies, 31 sites	Data held by DoE (NI), no public report
Coastal and marine areas						
M1	UK National (Marine) Monitoring Plan (UK NMP)	MPMMG	Organic & inorganic variables in water column, sediment, shellfish & fish	Data from at least 1988 SF: water 1-4/yr, sediment 1/yr, biota 1-2/yr	Approx. 100 sites in the upper, middle and lower reaches of estuaries, inshore and offshore coastal sites around the UK	Central database being developed No UK report, Data passed to the North Sea Task Force
M2	Water classification of estuaries	SOEnD, RPB NRA DoE(NI)	Use related descriptions, aesthetic, biological, bacteriological, and chemical conditions	Start year 1985 Every 5 year SF: 4/yr, variable in Scotland	All Scottish estuaries exceeding 1 km 28 estuaries in England and Wales All 7 N.Ireland sea loughs and estuaries	Database: SOEnD, NRA, DoE(NI) Reporting: SOEnD, NRA, DoE(NI)
M3	Classification of coastal waters	SOEnD, RPB	Use-related descriptions, aesthetic, biological, bacteriological, and chemical conditions	Start year 1990 Every 5 years SF: variable	Coastal waters of Scotland Approx 7,000 km length	Database: SOEnD, NRA, DoE(NI) Reporting: SOEnD, NRA, DoE(NI)

No.	Name	Responsible institution	Variables	Period of operation & Sampling Frequency (SF)	Geographical coverage	Data & reporting
M4	Marine Algae Monitoring Programme	NRA DoE(NI)	Marine Algae	Since 1991 Weekly from May to September	England & Wales: 615 identified and non-identified bathing waters Northern Ireland: 16 identified and 10 non-identified bathing waters	Summary data held nationally Annual internal report
M5	Monitoring of Bathing waters.	NRA, RPBs	Bacteria, organic pollution	20 times a year during the bathing season	460 bathing waters in England + Wales(421), Scotland(23), and N.Ireland(16)	Annual reporting
M6	Water Quality of Shellfish Waters.	NRA	Heavy metals, organic micropollutants	SF: Variable 2-12/yr	29 shellfish waters	Annual reporting

DoE: Department of Environment; NRA: National River Authority, England and Wales; SOEnD: The Scottish Office Environment Department; RPB: River Purification Boards, Scotland; DoE(NI): Department of Environment, Northern Ireland; MPMMG: Marine Pollution Management Monitoring Group;

Table 2.2-16 International inland surface water monitoring programmes (Kristensen and Bøgestrand, 1996)

No.	Name	Responsible institution	Variables	Period of operation & Sampling Frequency (SF)	Geographical coverage	Data & reporting
Rivers and streams						
EU-R1	EU river network. Exchange of information Council Decision no 77/795/EEC 1977	CEC and Member States	18 physical, chemical, and microbiological variables	Since 1977 Portugal and Spain from 1986 SF: monthly samples	Large rivers in the EU Member States 126 sampling sites	Database: CEC Reporting: every three years by the CEC
	Global Environment Monitoring System GEMS /Water	WHO & UNEP GEMS/WATER Collaborating Centre Canada	Major ions (7) Metals (12) Nutrients (3) Organic micropollutants (5) Basic variables (4)	Since 1977	60 countries, world-wide, currently participate in the GEMS/WATER programme and around 360 surface water sampling sites are included	
	OECD	OECD			Rivers in the member countries.	Reporting every 5 years by OECD
R-R1	Rhine	ICPRP	Water: 61 chemical and physical variables Suspended solids: 30 chemical and physical variables	Since SF: Water 12/yr to continious Suspended solids 12-24/yr	9 sampling sites on the main course of the river Rhine	Database: ICPRP Reporting annually by ICPRP
	Elbe	ICPE	10 heavy metals, 16 organic micropollutants and 5 biological variables		16 sampling sites	Database: ICPE Reporting: ICPE
	Danube Bucharest Declaration		Chemical and physical variables. Nutrients, heavy metals, organic micropollutants and petroleum products Specific surveys performed in 1991-92.	Since 1988	11 sites	

No.	Name	Responsible institution	Variables	Period of operation & Sampling Frequency (SF)	Geographical coverage	Data & reporting
Lakes and reservoirs						
	Lake Constance/Bodensee	IKGB	Eutrophication variables, oxygen, major ions, heavy metals, organic micropollutants, radionuclides Hydrobiological and microbiological variables (phyto- & zooplankton, bacteria)		Lake Constance 3 water sampling sites	
	Lake Geneva/Lac Léman	ICPGP	Water quality variables including eutrophication variables, heavy metals and organic micropollutants Hydrobiological and microbiological variables. Sediment monitoring		Lake Geneva 1 water sampling site & 200 sediment sampling sites	

CEC: Commission of European Communities; IKGB: International Gewässerschutz-Kommission für den Bodensee/International Commission for Protection of Lake Constance;
ICPGP: International Commission for Protection of Lake Geneve against Pollution; ICPRP: International Commission for Protection of the Rhine against Pollution; ICPE: International Commission for Protection of the Elbe

2.2.3 Issues with interpretation of results

The number of pesticides monitored varies depending upon size, geographic location and monitoring strategy throughout Europe. As an example, in England and Wales in 1996 a total of 147 pesticides were monitored by water companies and 163 pesticides monitored by the Environment Agency. In 1997 a total of 1419 sites were monitored for pesticides by the Environment Agency (PEWG, 2000). A summary of the pesticides most frequently exceeding 0.1 μg/l is provided based upon the Environment Agency monitoring programme for 1997 is summarised in Table 2.2-17.

Table 2.2-17 Pesticides most frequently exceeding 0.1 μg/l and environmental quality standards (EQSs) in surface freshwaters in England and Wales in 1997 (Environment Agency, 1999)

Pesticide	Total number of samples	% of samples > 0.1 μg/l	% of monitored sites failing any EQS
Isoproturon	3571	17.4	0.3
Mecoprop	3526	12.6	0.2
Diuron	3579	11.9	0.5
MCPA	2120	5.7	1.4
PCSD or Eulan	904	5.5	15.6
Simazine	6284	5.3	0
Atrazine	6409	4.6	0
2,4-D	2586	4.4	1.2
Oxamyl	784	4.1	N/A
Cypermethrin	1007	2.3	45.0
Diazinon	4317	2.2	13.1
Permethrin	1079	2.0	42.6
Carbofuran	1040	2.0	N/A
Carbaryl	1075	1.9	N/A
HCH Delta	2345	1.6	0.5 (total)
Aldicarb	947	1.5	N/A
Bentazone	1638	1.4	0
Dichloprop	1393	1.4	N/A
Propetamophos	3896	1.4	9.3
Chlorotoluron	3619	1.4	0
Pentachlorophenol	3870	1.1	0
Dichlorobenil	1300	0.9	N/A
Cyfluthrin	978	0.9	42.4
Alpha HCH	6424	0.9	0.5 (total)
TBT	1861	0.6	10.3

N/A: EQS not available

Because of resource limitations the interpretation of monitoring databases such as those summarised in Tables 2.2- – 2.2-16 and highlighted in Table 2.2-17 is not necessarily straightforward. The following issues need to be considered with care:

- Which monitoring programmes considered the pesticide in question?
- Were relatively simple screening methods employed to identify presence of chemical classes or were more complex methods used to confirm identity and for quantification?
- What size of water bodies was sampled?
- Were the analytical methodologies in use at the time allow for quantitative analysis at the levels required for the risk assessment?
- What was the usage and landscape context in which detections were found?
- Were any attempts made to track and identify causes for large-scale detections (i.e. point source contamination etc.)?
- What sampling strategy was employed in the monitoring programme and how well does this take into account the primary route of entry?

This final point needs to be considered with great care when making use of water monitoring programmes. This defines not only the spatial frequency of sampling but also the temporal frequency and any response to hydrological change. For example, if the primary route of concern is drainage or run-off, monitoring programmes including a rainfall-response sampling strategy is obviously important. It must be recognised when interpreting existing databases that this aspect may not be included in the programme design as sampling strategy is often more simplistic.

Unfortunately, it is difficult to answer all of these questions with historical databases with certainty. This can impose limitations on the value of such data within updated product risk assessments. However, the availability and quality of data are constantly improving and new spatial techniques are providing tools to assist in the interpretation of this information. Where robust and detailed databases exist, there is the potential to make very effective use of this information to provide practical demonstrations of actual exposure potential associated with real usage situations. Where sufficient water quality and hydrology data are available monitoring exercises can provide a useful database for estimating parameter ranges required for modelling predictions and facilitate comparisons with other (real or simulated) systems.

2.2.4 Potential impact of point versus non-point sources

Large scale water monitoring programmes such as those described earlier cannot differentiate between exposure arising as a result of spray drift, run-off or drainage following recommended agricultural practice and that arising from accidental spills or other large or small point sources. Yet point source contamination of surface waters from pesticides within agricultural catchments can be significant. In some cases, contamination of surface waters via point sources can be as great or greater than diffuse sources. Critical point sources include areas on farms where pesticides are handled, fill into sprayers or where sprayers are washed down (Rose *et al*., 2001).

Monitoring projects conducted in the UK (Mason *et al*., 1999) and Sweden (Kreuger, 1998) have identified that point sources can be responsible for a significant proportion of the total amount of pesticide loading in surface waters and can account for the peak concentrations detected. For example, reported ranges run from at least 20% of the total loading up to as much as 70%, depending upon catchment characteristics.

A case study in the UK based upon the River Cherwell (Mason *et al*., 1999) considered the origin, timing and magnitude of losses of isoproturon (IPU), a pesticide for which drain flow is a significant route of entry. In this study water monitoring data was obtained from the regional water supply company and was subjected to careful interpretation. It was demonstrated that monitored IPU concentrations rose prior to the period when land drains in the catchment were known to have started flowing. This implied that other agricultural operations, including point sources, provided a more significant contribution to surface waters than had been previously realised. The results of a more intensive monitoring study based in the same catchment have demonstrated that point source contamination is generic and not limited to specific chemicals.

In Bornholm, DK, 10 agricultural enterprises were investigated. Water samples were taken in secondary groundwater close to the soil surface. Samples were taken in areas adjacent to the farm and not in the treated fields. The total concentrations of pesticides measured ranged between 3 and 1720 μg/L with peak concentrations of individual pesticides reaching 800 μg/L. The high concentrations were assumed to be caused by excessive pesticide use, inappropriate washing locations for spray equipment or inappropriate handling of pesticides (Bay, 2001).

In a review of 42 investigations on 40 sites where pesticides are sprayed for farmers, the picture is similar (Amternes Videncenter, 2002). Pesticides were detected on 94% of the sites investigated. It is thought that there are more point sources with large concentrations that the study indicated.

Kreuger (1998) found that atrazine, hexazinone, propyzamide, simazine and terbutylazine (and to some extent bentazone and cyanazine) detections resulted from applications on non-agricultural land such as farmyards. In a single case, runoff from a farmyard resulted in a stream concentration of 100 µg/L. In total, more than 6 kg of terbutylazine was washed from the land surface over a period of seven months. Large concentrations of a number of pesticides were detected in groundwater collected from farmyards with significant leaching occurring for several months. Furthermore, two cases of spills resulting from filling or cleaning of application equipment were identified. The author concludes, "Indeed, a substantial contribution of pesticide loss to stream water was from the application of pesticides in farmyards."

German results (Müller et al., 2000) have shown that over 75% of the pesticide loading in a monitored stream originated from a storm sewer which transported farm runoff to the stream. In the final empirical catchment model developed by the authors, only the application rate of pesticide applied in the fields was a significant variable. The measured stream concentrations ranged between 0 and 23 µg/L.

2.2.5 Use of data for ecotoxicolgical assessments versus drinking water evaluations

As summarised earlier, water monitoring programmes are established with a very wide variety of objectives. These objectives shape the designs of the studies, but in doing so, also potentially limit their wider application as environmental risk assessment tools. A summary of the general advantages and disadvantages of monitoring data are summarised in Table 2.2-18 based upon a review undertaken by ECOFRAM (1999).

Table 2.2-18 Summary of advantages and disadvantages of use of monitoring data within risk assessment exercises (After ECOFRAM, 1999)

Advantages	Disadvantages
Provides an actual measurement of chemical residue concentration, hydrologic response etc	Costly
Accounts for the inherent heterogeneity of the system	Time involved is weeks to years
There is a greater acceptance of measured data	Difficult to design cost effective AND technically viable sampling programs
There is public confidence in monitoring data	May require many years of monitoring and/or paired studies to evaluate effectiveness
	Handling non-detects can be problematic
	Results are accepted as 'true' values without necessarily understanding context
	Sampling represents discrete points in space and time
	Study only represents one unique combination of conditions
	Can be constrained by analytical precision and limit of detection
	Results can be misleading if one year monitored is a 1 in 100 event year
	Cause and Effect may be difficult to assign

The conclusion of the analysis undertaken by ECOFRAM (1999) was that '*monitoring can usefully be thought of as another "model" with definable but relatively high uncertainties*'. In many cases the most significant uncertainties are associated with the potential of sampling timings to capture an "event" of ecological significance. This highlights a clear divergence in the temporal objectives of water monitoring as applied towards evaluation of drinking water resources and as applied within ecological risk assessments. Broadly, the purpose of drinking water monitoring is to pick up background contaminants present in water prior to abstraction. The focus is generally upon chemicals that may be present in water supplies over extended periods of time and will not focus on transient pulses of chemicals. In this respect, drinking water evaluations may provide useful information for chronic exposure assessments, but the ability to accurately capture critical acute exposure periods is literally 'hit or miss'.

Among potentially the most powerful use of thoroughly planned monitoring studies conducted over several years that has been suggested by ECOFRAM (1999) are as a means of calibrating models to increase regulatory confidence. Such modelling could provide probabilistic estimates of exposure across time and space to set the monitoring data into context by consideration of the actual rainfall experienced and the watershed(s) involved.

Under some circumstances, even though a formal validation may not be possible, the generalised use of monitoring data may be used to provide an overall increased confidence about other pesticide transport assumptions and/or modelling approaches that could then be used to support ecological risk assessments.

2.2.6 Acceptance/rejection criteria for monitoring results

On the basis of this overview, a number of acceptance/rejection criteria can be established based upon the general uncertainties outlined earlier. Initially, the relevance of the programme needs to be established:

- Did the monitoring programme consider the pesticide in question?
- Were the analytical techniques used suitable for characterising the chemical in question at levels of ecological significance?
- What was the agricultural context of the catchment being monitored (relevance of cropping to product usage)?

In addition, the following, often less easily determined aspects need to be considered:

- Are large-scale detections attributable (i.e. point versus non-point sources etc.)?
- How does the timing of these large-scale events compare with the anticipated primary routes of entry (i.e. do significant loadings coincide with rainfall events, initiation of drainflow periods in the autumn, periods of expected usage)?
- What sampling strategy was employed in the monitoring programme and how well does this take into account the primary route of entry (i.e. sampling coinciding with rainfall events, sampling during intensive usage periods etc…)?

Since the purpose of higher tier risk assessment is primarily to reduce uncertainty the use of such databases as risk assessment tools should be considered with great care. Ultimately, the results of an individual monitoring programme can be a very powerful risk assessment tool, but only if a sufficient number of these critical uncertainties can be clearly overcome.

2.2.7 Design of surface water monitoring programmes

As summarised earlier, monitoring programmes may be conducted for a variety of reasons. Of specific relevance to supporting environmental risk assessment submissions for regulatory evaluation the following could be considered:

- To support higher-tier field research (drainage studies, run off studies etc…),
- To investigate presence of chemicals not considered by routine monitoring described above (specific use situations, recent registrations etc.),
- As components of product stewardship campaigns.
- To assist in calibrating modelling assessments of exposure

A summary of characteristics of monitoring programmes conducted at three illustrative scales are summarised in Table 2.2-19. The key component of any monitoring programme is the sampling and analysis strategy. The strategy developed is determined by the objective and scope of the study, campaign or research programme that it supports. The key considerations are:

- Is the chemical likely to be detected (is it used?)
- Number of samples and frequency
- Method of sample gathering
- Location
- Number and type of determinands per sample
- Level of detection
- Efficiency of recovery
- Cost
- Time

Location is probably the most important design consideration. Wauchope and co-authors (1995) have concluded that if the study is to address risk assessment concerns or to monitor use of many pesticides, the study area should be as large as possible. The use of a very large area maximises the chances of detecting traces of a chemical which may occur only in a single water body and will be missed if that particular water body is not sampled. A range of agricultural conditions may also be covered (e.g. topography, hydrology).

Wauchope and co-authors (1995) have pointed out that unless an automated or response-driven sampling strategy is developed the practical limit on size is likely to be the number of samples which can be collected in a reasonable time period and travel between sampling points. On the basis of the experiences of Frank *et al*. (1982) it was concluded by Wauchope *et al* (1995) that where this is not possible a sampling to support a run-off monitoring programme based upon twelve locations in an area of ca 15 kilometres square can comfortably be sampled for water and sediment in a working day.

Table 2.2-19 Summary of Design Considerations for Monitoring Programmes (based in part upon ECOFRAM, 1999)

Design Consideration	Small-Scale Test Plots	Sub-catchments	Catchments
Drainage area size	<0.05 HA	10-40 ha	10 to > 100 km^2
Flow regime studied	Overland (partial)	Overland, ephemeral streams, ponds	Perennial streams, rivers, lakes, reservoirs
Point of interest	Localised runoff, drainage or drift	'Worst-case' exposure	Large scale exposure and dilution
Site characterisation	High	Moderate/high	Low
Control over system	High	Moderate	Low
Simulation of precipitation	Yes	Difficult	No
Study duration	Days to years	Seasons to years	Years
Field heterogeneity	Neglected	Represented	Represented
Field-scale influences on pesticide transport	Neglected	Represented	Years
Focus	Research, idealised system, label use	Label use	Reality

The field instrumentation used to support the study design needs to take into account all of the points described earlier. Water can be sampled by a wide range of techniques and devices in order to collect either bulk samples (e.g. from the 'centre' of a water body), from the surface (where hydrophobic chemicals may concentrate) or including both the surface layers and depths (integrated-depth sampling). Instrumentation that supports an automated sampling strategy provides an opportunity for developing customised sampling regimes. However, there are practical limitations that need to be recognised. Power failures and fouled sampling equipment are common. Sediment sampling also represents a very significant technical challenge. Very large variability is often encountered in sediment sampling and can create heavy sampling burdens and interpretation problems. A variety of passive water sampling techniques have been proposed recently that hope to avoid some of the more problematic aspects of monitoring study design and interpretation. Ultimately, however, Hendley (1995) has concluded that there is no substitute for having a person on-site during critical periods in a monitoring study to help ensure that representative samples are taken. This view is shared by Wauchope *et al*. (1995) in their review of methods and interpretation of run-off field studies.

The UK Environment Agency (in collaboration with the Scotland and Nothern Ireland Forum for Environmental Research (SNIFFER)) and the Po River Authority in Italy (with scientific support of the National Research Council – Water Research Institute) have developed a detailed Manual of Best Practice for the design of water quality monitoring programmes (Environment Agency, 1998). The manual has the purpose of supplying step by step guidance to organisations responsible for water monitoring activities on what, how and when to sample and how to analyse the resulting data and generate management information. Consultation of this document would be of obvious benefit when considering the development of large scale monitoring programmes.

2.2.8 References

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2.3 Landscape analysis

The concept of refining Step 3 modelling input parameters using spatial approaches is an important component to Step 4 approaches, and will be introduced in several sections of this document. The ability to efficiently analyse large amounts of spatially related information pertaining to the agricultural / aquatic relationship has broadened the horizon beyond Step 3 scenario parameters when needed in the risk assessment process. This involves combining and processing of harmonized thematic data sets within a Geographic Inforamtion System (GIS) to gain a refined (and quantitative) understanding of the agricultural landscape. The data sets include information on environmental conditions, agricultural practices, statistical input and ecological indicators as well as information derived from remote sensing instruments.. In addition to the refinement of Steps 3 inputs and development of new modelling scenarios, landscape analysis can also provide inputs for other higher tier modelling methods, such as catchement-scale and probabilistic approaches.

Topics covered within this section include: a discussion on the spatial unit of analysis, the site selection process, discussions on a number of landscape factors that can be used to describe the agricultural landscape, several examples of calculating exposure estimates using spatial approaches, relating landscape factors to a larger area, other supportive information for higher tier exposure assessment, and the use of remotely sensed data in landscape characterization.

While attempts have been made to compile a cross section of spatial approaches, the scope of this task is large, and the reader should also review literature for additional approaches.

2.3.1 Unit of analysis

For the purposes of surface water aquatic risk assessment, there are several approaches to examining the “unit of analysis”. The unit of analysis is that spatial feature to be examined as either a contributor or receiver of potential exposure. In simple terms, this can be either the water body (as that unit receiving potential exposure and which should be evaluated), or the agricultural field (as that unit contributing potential exposure, with possible mitigation). Other approaches include the analysis of catchment areas, grid cells, and even individual water body segments. All approaches are valid for specific purposes, and all should be considered for the specific application. Data requirements, data availability, level of GIS complexity and processing time required are all considerations that should be taken into account.

2.3.1.1 Agricultural field

Field level approaches might be used when the goal is to identify those agricultural areas that may contribute the greatest amount of potential exposure, with the goal of regulating pesticide applications or mitigation efforts. In many ways, this approach makes most sense to the farmer and may be the best implementation of location specific mitigation measures. In other words, using the field as the unit of analysis provides the ability to selectively implement a variety of mitigation measures based on the potential exposure of those fields (type of exposure, temporal aspects, other neighboring inputs, etc.).

The definition of a "field" should be considered before the application of this type of analysis. Many land cover data sets do have field boundaries based on land parcel information (e.g., topographic data sets such as the ATKIS DLM/25 in Germany, crop subsidy management systems, etc). In some cases, where land cover information is generated from remotely sensed data (satellite or aerial imagery), the delineation of fields (as opposed to simply cropped areas) should be clarified up front, so that it can be implemented if possible. Using satellite imagery of 10 to 30 metres in resolution, it is unlikely adjacent fields of similar crop types can be separated.

2.3.1.2 Water body

Water body level approaches to landscape analysis have the appeal that they represent the structure in which surface water protection goals are aimed. There is also great diversity in water body characteristics and potential exposure, resulting in a distribution of exposure results, rather than a single value. These results can be used as inputs to catchment-scale modelling and probabilistic approaches. An understanding of all impacts to a water body, regardless of field definitions, allows for a more holistic approach to exposure assessment, as multiple inputs can be examined.

As with the field approach, the working definition of a water body is important to understand when assessing the results. Hydrologists and risk assessors may have different views on how a water body is defined (based on physical/morphological characteristics, or on exposure potential). Commonly a flowing water body in a GIS is defined as the length between one confluence and the next, or the headwater to first confluence. This provides a consistent morphological water body (i.e., flow is unlikely to change without inputs from confluences), and is consistent with some stream ordering methods, but can result in streams that may be longer than appropriate from a risk assessment point of view (i.e., larger than the 100-metre edge-of-field water body at Step 3). Static water bodies such as ponds are already spatially distinct and do not need much discussion in this area.

One approach to the dual view of a water body is to use a two-tiered approach. This method uses the "confluence to confluence" method as the first tier, producing hydrologically relevant exposure results, and also sub-dividing these water bodies into segments of equal and exposure-relevant lengths (e.g., 100 metres). While more difficult to implement, the results are more usable for both current and future needs (i.e., re-cast results using different segment lengths, or update exposure results for the "confluence to confluence" water bodies with newer physical characteristics).

It should be noted that when GIS-ready hydrology data is used in an analysis, many times a unique water body identifier is already present from the data provider. How this water body identifier was assigned should be investigated. In the least, the source water body identifier should be maintained throughout the spatial processing so final exposure results can be related back to the source data for comparison or correlation with other factors from the data provider (or other organizations using the same set of base hydrology data).

2.3.1.3 Catchment

The use of catchment areas as the unit of analysis is suited for issues related to spatial factors beyond the field-to-water body analysis. While all units of analysis encompass a speficic area on the ground, catchments are hydrologic units, unlike agricultural fields or grids (i.e., a 10 x 10 km area). They represent a larger area (though they can vary in size/scale) than water bodies, and can be a unit of analysis individually or used as a method for ranking relative vulnerability over larger areas.

Evaluating the potential exposure for individual catchments can incorporate cumulative exposure beyond the field to water body method. The complexity of catchment level analysis can vary from simple exposure factors for the entire catchment, to very complex dynamic factors such as movement within the hydrologic system, temporal aspects and more complex spatial relationships between environmental factors and the surface water.

2.3.1.4 Grid cell

The grid (i.e., a series of rows and columns of equally sized cells; such as a 25 x 25 km grid across the EU) as a method to evaluate potential exposure can be used at a variety of scales. The grid approach as a unit of analysis generates landscape factors for each grid based on the underlying spatial data (e.g., total length of flowing water in a 10 x 10km area). A formula or function may also be used to combine landscape factors into an indivudal exposure metric for each grid cell (e.g., exposure = [crop density * rainfall * soil factor] each 10 x 10km area). The size of the grid cell (e.g., 25 x 25 km, 10 x 10km, 1 x 1 km, etc.) should be chosen as to be appropriate to the area of study, the issue under examination, and the landscape data available. Small grid cell sizes (i.e., 5km x 5km) should be used to study smaller areas, where available landscape data are more spatially refined, or to gain greater granularity in the spatial distribution of the results. Larger grid cell sizes (25 x 25 km) can be used to cover larger areas (specifically for the site selection and/or vulnerability ranking process aimed at refining the area of analysis for futher, more detailed, analysis), or where data sets do not allow for greater resolution.

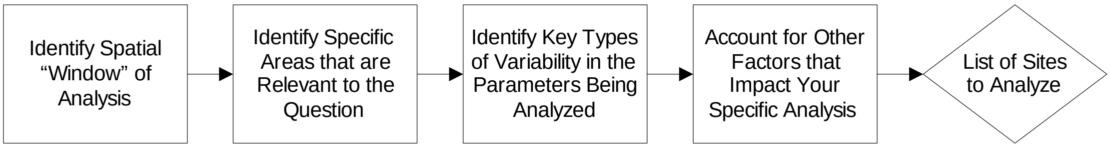
2.3.2 Site selection process

The landscape analysis techniques and subsequent results described hereafter, while valuable tools in and of themselves, realize their full potential only when placed within a greater spatial context. By identifying regions that are considered to be homogeneous with respect to the parameters that are being analyzed, the location and number of individual study sites necessary to capture the variability inherent within the area of analysis can be determined. The site selection process allows the analyst to then make statements regarding a larger area based on an analysis of a subset of that area.

This process is especially important when Step 3 scenario parameters are going to be refined. It is important to start with the extent of the EU in which the scenario is representative, so that when one or more modelling parameter is refined due to landscape-level analysis, there is confidence that the results are indicative of a typical to highly vulnerable area for that scenario.

2.3.2.1 Generic methodology for site selection

There are many ways to conduct a site selection for landscape-level analysis, but in general a generic framework can be followed.



The generic framework above can be represented as a number of questions that should be answered in order to determine where and how many sites might be required:

1) What is the spatial "window" of the analysis? Is the analysis to cover the entire EU? A Step 3 scenario? A single country? A specific region that exists across national boundaries?

2) Within this spatial window, what specific areas contain the factors to be analyzed? For example:
   - Where are the crops of interest grown?
   - What are the predominant soil and climate conditions?
   - Where do the types of water bodies of interest occur?
   - Where do the species of interest exist?

3) Is there any "sub-variability" in the factors to be analyzed that should be accounted for? For example:
   - Are there different methods of application?
   - What are the important transport processes?
   - What types of water bodies present?
   - Are there different habitats for a species of interest?

4) Are there other factors that may impact the analysis? For example:
   - Availability and accessibility of data (or lack thereof)
   - Political factors
   - Financial considerations

By answering these questions, it is possible to:

- Determine with some level of confidence the location (and number) of study sites;
- Represent the variability with respect to the parameters to be analyzed; and,
- Place each study site into a larger context that identifies the spatial extent to which the results can be extrapolated.

2.3.2.2 Scale in the site selection process

In order to better understand the site selection process and how scale plays a key role, it is instructive to use a real-world example. Annex A4 of Volume 1 describes an example landscape analysis that was conducted for citrus. In short, the goal was to identify an area that adequately represents citrus production within the EU (see the text of the example for more details). In order to perform the site selection, a series of analyses were undertaken representing different scales of information from macro to micro until a specific area could be identified and selected for analysis.

As stated above, the process can be broken down into a generic framework that was followed in this example. The first step is to define the spatial "window" for analysis; in this case simply the EU as was defined in the problem statement. The next step is to identify specific areas that are relevant to the desired analysis. For this example, we are interested in citrus, so citrus production within our spatial "window" (the EU) was examined. To do this, agricultural statistics were used to identify the countries that produced citrus, and as the example indicates, Spain accounts for the largest single portion of the citrus production. To this point in the process, things have been looked at in the coarsest *spatial "window" scale* using EU-wide statistics. This is often the starting point for a site selection process, and leads to a refinement of the analysis. Regions of importance for specific crops with very restricted spatial extent (e.g., Altes Land in Germany) may not be found by starting from relatively general data at the EU level, and may require more targeted examination baed on more specific local knowledge. However, the results of the landscape analysis performed on such a specific area may be more difficult to extrapolate to other areas and to provide relevance for Annex 1 examination.

Looking at our flowchart above, we move to the next step which is identifying specific areas within the spatial "window" that are relevant to the question. In order to do this for our example, additional *national scale* cropping statistics were used to identify areas within Spain that would be candidates for analysis. As can be seen in Figure A4.1 of the example, the Valenciana region in eastern Spain stands out as the primary production area for citrus within Spain.

In some cases, narrowing down our site to this point may be sufficient, but because landscape analysis often requires that the analysis be performed on individual water bodies or other relatively detailed units of observation, it may be necessary to further refine the selection of areas. In this example, we move beyond a national scale to a *regional scale* through the use of even more detailed crop statistics describing smaller spatial units (*municipios*). As can be seen in Figure A4.3 of the example, specific areas within the Valenciana region are now identified as primary candidates for our detailed landscape analyses. The results of the detailed crop statistics were then compared to the distribution of tree crops based on the CORINE land cover.

By working down to this regional scale, we have identified a specific area within the EU that is manageable in size for a detailed analysis of the landscape interaction between citrus and water bodies. We can also feel confident that because of the intensity of the production, we will be analyzing a statistically significant sample that is representative of citrus production within our spatial "window". Once we have achieved this goal, a specific area of analysis is then identified based on a unit of analysis (discussed further in the next section). For this example, the unit of analysis was the footprint of a satellite image that will be used to quantify the landscape within the study area. As can be seen in Figure A4.4 of the example, a single satellite image will encompass the area of interest and that area of coverage will be used to define the final *study area*.

2.3.2.3 Units of analysis

As described above, scale is an important factor in the site selection process and it is generally necessary to solve the problem in an incremental fashion moving from macro to micro scales of analysis. Often, it is difficult to determine the units of analysis that can or should be used as one moves through these different scales. Below are some examples of units that may be appropriate for helping to quantify the site selection process:

- Because landscape level analyses often involve agriculture, *National or Regional Crop Production Statistics* can be used to identify sub-regions (nations) within the EU that produce the crops of interest and eliminate those areas that are not of concern.

- The analysis may involve issues or factors that do not lend themselves to discrete units of analysis that are arbitrary such as national boundaries. In this case, it is often more appropriate to use *landscape characterization units* such as EcoRegions or other similar environmentally based spatial units. EcoRegions are areas of similar environmental composition with respect to climate, landscape (and sometimes biologic) factors.

- Other environmental units such as *hydrologic catchments* (described in Section 2.3.1.3 above) may be appropriate when hydrologic phenomena are being analyzed. Catchments can vary in scale from macro (major fluvial systems) to micro (individual water body catchments). As a spatial unit of analysis for site selection, catchments can be extremely useful for delineating areas of interest.

- Often an analysis is driven at least in part by political or legal concerns. In these cases, it may be necessary to use some form of *administrative units* to delineate areas of analysis in the study site selection. These may be member state boundaries or sub-national boundaries such as provinces or cantons, etc. and often some form of administrative unit can be used that is appropriate for the scale of analysis being conducted. As an example, macro (member state), regional (provinces) and micro (*municipio*) boundaries were used in the citrus site selection described above.

- Other more arbitrary units may be used as appropriate, perhaps driven by the format in which the underlying data is provided. For example, in the citrus site selection a final study area defined by the footprint of the satellite imagery used to provide the landcover information for the study was used to define the final study sites. In general, however, the spatial unit of analysis should be driven by the question being answered and not the other way around.

2.3.2.4 Data consistency

One important consideration when performing a site selection analysis is to account for differences in data that may be used. It is always desirable to use some form of data that provides the same type and detail of information across the entire spatial “window”. For example, let us assume that hydrologic density is a factor to be considered in a site selection process. Using a dataset for one region (member state) within a spatial “window” that provides detailed spatial information about the smallest water bodies and then using another dataset for a different region (member state) that does not provide the same level of spatial detail and omits the smallest (but most common) water bodies can lead to biases in the site selection process. In this case, the more spatially detailed data would create the appearance that those areas covered by the dataset were more hydrologically intense, while this may not actually be the case. Because of these types of factors, it is imperative that the analyst take the level of detail inherent in a data set into account when conducting a site selection analysis.

2.3.3 Landscape factors

A more refined understanding of the agricultural landscape can be obtained with available geodata and spatial technologies. This section will discuss a number of landscape factors that can be derived from spatial data, and that can then be used to refine model inputs for Step 4 analyses. It should be noted that the landscape factors presented are concepts, and should not related to a specific methodology or data set. There are numerous ways to achieve similar end results via differing data sets and methods. While example methods are presented, they do not represent the only method that is appropriate. As in all analyses using spatial data and approaches, a complete description of the input data sets and processing methodology should be provided in order to interpret the results appropriately. The landscape factors discussed in this section are outlined in the following table. Each area is discussed briefly in beginning of this section, followed by a more exhaustive description later in the section.

Landscape Factors related to:
Meteorological and climate inputs
Soils
Catchment area and characteristics
Elevation and derived metrics
Hydrologic density
Cropping density
Buffer width
Buffer composition
Perimeter composition
Proximity of crop to water
Wind speed and direction

Meteorological information is a key element in estimating and refining exposure to surface water using existing models. Data that are more temporally comprehensive or more spatially refined can be used in Step 4 to refine the parameters used at Step 3 to be more representative of the specific conditions, crops, etc. being examined. Spatial distribution estimates of meteorological data are very important especially as inputs to spatially explicit landscape, regional, and global models. Nevertheless, the most common sources of climatic data are meteorological stations, which provide data only for single and sparse locations (discrete points in the space). Methods for interpolation of point data are discussed.

The characteristics of soils play a major role in the potential exposure of surface water from runoff and drainage routes. A number of soil properties can be important in the landscape level evaluation of soil-related exposure, including soil texture, pH, hydrologic group, organic carbon content, etc. Soil attribute data that are more comprehensive or more spatially refined can be used in Step 4 to refine the parameters used at Step 3 to be more representative of the specific conditions, crops, etc. being examined. An understanding of the organization of tabular (attribute) and spatial soil data, using the Soil Geographical Data Base of Europe as an example, is the focus of this section.

Catchments are commonly desirable as a unit of analysis for surface water, specifically for drift and surface runoff issues, since they can be used to spatially relate land area, and other characteristics, to surface water. The catchment area, cropping patterns and treated amount are parameters used in the Step 3 scenarios. Refinement of these parameters is possible using landscape-level data. Larger catchments can also be used as a unit of analysis for relative vulnerability studies in which large areas are examined to determine a suitable detailed area of study that can be ranked and/or quantified. The Catchment Characterization and Modelling data set (from the Agri-Environment Action managed by the Soil and Waste Unit of the JRC Institute for Environment and Sustainability) is discussed as a possible data source.

Elevation and derived metrics such as slope, aspect, flow accumulation, and compound topographic index (CTI) can be useful in the estimation of surface water and pesticide runoff. These metrics can also provide useful landscape characteristics such as the potential presence of flowing surface water (can be used in the absence of available vector hydrography), surface water flow direction, and drainage area. Refinements in slope and drainage area have direct implications with respect to Step 3 model parameters (at the local level), while broader use of coarser-elevation derived metrics provide inputs for relative ranking of potentially vulnerable areas across the EU for Annex 1 investigations.

The hydrologic drainage density of an agricultural area may be an indicator of relative potential exposure to surface water from agricultural pesticides. Drainage density is calculated as an indicator of the average drainage capacity of the hydrographic network over a certain area. The relative ranking of hydrologic drainage density can be used to compare regions (both within a single MS and between MSs, or perhaps in the development of new scenarios), and can also be an important part of the broad site selection process, in order to ensure that the final area studied represents the level of agricultural / hydrological interaction desired.

Estimation of cropping density is a common modeling parameter to refine as it is relatively straightforward to estimate and provides an understandable and transparent modification to the Step 3 scenarios. There are a number of methods that can be employed to derive cropping density, depending on the unit of analysis, available data and spatial processing complexity.

A primary factor in assessing potential exposure to surface water is width (and composition) of naturally occurring buffers between cropped area and surface water. These buffer areas may supply some natural mitigation to spray drift or the runoff process. In order to quantify the buffer width and composition (when it exits), it is important to define the spatial component of the buffer. The width of the naturally occurring buffer is a direct parameter in the Step 3 models, and provides an easily understood method for modifying Step 3 scenarios in the Step 4 process. While the concept of a buffer is equally valid for both drift and runoff, the characteristics of the buffer that promote mitigation are different for the two processes.

The composition of the water body perimeter describes the relative proportions of perimeter that are in different exposure categories (directly adjacent, buffered, and non-cropped). This may be important because the spatial relationship of a water body to crop (the potential exposure) varies along the length / perimeter. This measurement provides an indication of the variability of potential exposure along the water body perimeter, beyond simply a single PECsw value. While this is not a standard parameter in Step 3, knowledge of this exposure variability can enhance the understanding of the influence of the landscape related to exposure.

The crop proximity metric provides a single value to describe the distance between crop and the water body (as opposed to the buffer width which only describes the buffered portion of the water body, not the entire water body). One method combines both the measured distances to crop along the perimeter, with the percentages of water body perimeter that fall within specific classes, into a single index. This crop proximity metric provides a single value that expresses extent of potential exposure (as % of perimeter exposed) and the intensity of the exposure (as distance from crop to water). Another approach uses the agricultural field as the unit of analysis, and determines the portion of the field that is closest to the water body, and provides a (minimum) proximity value for the field based on that portion.

Using landscape-level information, the potential exposure of surface water to spray drift can be examined based on wind speed and direction, allowing for a refined examination of drift, possibly including alternate drift curves or exposure estimated where drift can occur from only one direction at a time during an application. A discussion on the general use and interpretation of wind speed and direction is presented.

The landscape factors summarized above are presented in more detail in the following sections.

2.3.3.1 Meteorological and climate inputs

The availability of spatial information about meteorological data and climatic conditions is an important factor for many environmental disciplines and natural resource management activities that use these variables as a basis to understand the processes they study. Spatial distribution estimates of meteorological data are very important especially as inputs to spatially explicit landscape, regional, and global models. Nevertheless, the most common sources of climatic data are meteorological stations, which provide data only for single and sparse locations (discrete points in the space). Meteorological stations collect various environmental data from a variety of instruments.

Meteorological information is a key element in estimating and refining exposure to surface water using existing models. Meteorological data vary in scope from global down to local level, usually with variation in the content, period of sampling, and spacing of stations. Of particular and direct interest to Step 3 modeling are the precipitation and temperature measurements, while other meteorological information (such as wind speed and direction) may also be applied at Step 4. Meteorological data that are more temporally comprehensive or more spatially refined can be used in Step 4 to refine the parameters used at Step 3 to be more representative of the specific conditions, crops, etc. being examined.

Both the extent of coverage and the period of coverage are important factors to consider when using meteorological data. One of the most useful databases is the meteorological data from the Monitoring Agriculture with Remote Sensing (MARS) program (MARS, 1997) This data set contains historical daily weather observations from several hundred meteorological stations across Europe from 1975 – 2003 (depending on stations). The spatial extent of this data also includes coverage of the new EU member states. The data are interpolated to a 50 x 50 km cell grid structure. (A complete description of interpolation techniques is included later in this section). The MARS data were used to define the agro-climatic zones as part of the Step 3 scenario definitions by the FOCUS Surface Water group (FOCUS, 2002).

It should also be noted that MARS data are uncorrected values, i.e. there is no correction for wetting losses, temperature, wind speed (leading to rainfall not being perpendicular to the measuring devise), effects of rain intensity or snow (which also whirl around in the air rather than drop perpendicular to the measuring device). In addition, the correction factors will depend on the type of rainfall station used. The correction factors are particularly large in in areas with significant annual snowfall. As an example, standard correction factors (%) calculated for the period 1961-90 for Denmark are shown below (Vejen et al., 2000).

Type	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Year
A	29	30	26	19	11	9	8	8	9	10	17	29	16
B	41	42	35	24	13	11	10	10	11	14	23	37	21
C	53	53	45	29	16	13	12	12	13	17	29	48	27

*The types A, B and C refer to different degrees of sheltering of the rainfall station

As the Danish rainfall varies between 550 and approximately 1000 mm/year, the correction amounts to100-200 mm/year. . The potential to underestimate rainfall should be carefully considered before using the MARS data set, possibly with coordination from the data providers, in order to use the data appropriately. Other sources of information on this issue include Allerup et al. (1997), Aune et al. (1985), WMO (1982, 1998), and Yang et al. (1995, 1999).

For some applications, rainfall intensity is an important factor, and daily rainfall data is not sufficient. This is most relevant for runoff estimation since the time period over which the precipitation occurred has a direct impact on the amount of surface runoff. Bissonnais et al. (Bissonnais, 2002) used the seasonal frequency of rainfall exceeding 15mm per hour as part of an estimation of erosion risk for all of France. National precipitation data from Météo-France was used in this analysis. Note that the threshold of 15mm per hour was determined to balance precision of the data and ability to provide discrimination for regions in France, and is not necessarily appropriate for other areas.

A station could collect data such as precipitation, temperature, pressure, relative humidity, wind speed and direction. Data are collected from meteorological stations on a periodic basis. The collection can be either manual or automatic. The parameters collected at the meteorological stations can be used in a variety of environmental studies including the development of climatological databases and modelling activities. The spatial array of data at each single location may enable an estimation of the value of properties at un-sampled sites (spatial interpolation). The value of a property between data points can be interpolated by fitting a suitable model to account for the expected variation.

Spatial interpolation is especially important in hilly or mountainous regions where data collection is sparse and variables may change over short distances. Interpolation techniques for meteorological variables evolved a lot during the last years thanks to the increased power of computers used for forecasting models. Traditionally, the applied method has been the linear interpolation between stations and the drawing of isolines based in the researcher knowledge of the studied area. But the more recent works are interested in searching statistical or mathematical relationships between geographical variables (orography, continentality, etc.) and climatological variables.

There are emerging techniques that use GIS (Geographical Information Systems) to model these climatological variables. GIS has today evolved into powerful management tools used for capturing, modelling, analysing and displaying spatial data and they represent an amalgamation of database technology with computer assisted cartography (Worboys, 1995). Analysis is achieved across data layers in an object orientated programming environment allowing spatial variables to be mathematically combined and statistically compared and thus producing new spatial datasets. Cimatological and meteorological phenomena are spatially distributed variable and hence GIS represent a useful solution to the management of vast spatial climate datasets for a wide number of applications. A combination of GIS and remote-sensing techniques may represent a very powerful solution to obtain and combine geographical variables and climatological variables.

Remote sensing enables the acquisition of large-scale comprehensive datasets where as GIS provides a means to display and analyse the data. For example, Digital Terrain Models (DTMs) can be manipulated in a GIS to provide a baseline climatological dataset. Traditionally these were derived using land-surveying techniques but are now remotely determined using Synthetic Aperture Radar. Comprehensive raster climate datasets can also be inferred from satellite imagery. For example, Schadlich et al (2001) produced land surface temperature maps by combining a DTM with brightness temperatures derived from

METEOSATs thermal infrared channel. Similar approaches have been used by J.Verdebout (2000) and El Garounani (2000) to generate surface ultra-violet maps of Europe and evapotranspiration maps of Tunisia respectively.

As mentioned before, sampled meteo-climatic data are point source in nature because the most common sources of climatic data are meteorological stations, which provide data only for single and sparse locations. One of the biggest challenges facing meteorology is the extrapolation of point climate data across a wide spatial domain through the interpolation of point station data across the landscape by geostatistical techniques.

In general, interpolation is a method or mathematical function that estimates the values at locations where no measured values are available. Spatial interpolation assumes that the attribute data are continuous over space. This allows for the estimation of the attribute at any location within the data boundary. Another assumption is that the attribute is spatially dependent, indicating that the values closer together are more likely to be similar than the values farther apart. These assumptions allow for the spatial interpolation methods to be formulated.

It has been shown by different works that there is no single preferred method for data interpolation. Aspects of the algorithm selection criteria need to be based on the actual data, the level of accuracy required, and the time and/or computer resources available. Selecting an appropriate spatial interpolation method is key to surface analysis since different methods of interpolation can result in different surfaces and ultimately different results. Statistical techniques can be used to evaluate the interpolation methods against independently collected data.

One of the simplest techniques is interpolation by drawing boundaries, for example Thiessen (or Dirichlet) polygons, which are drawn according to the distribution of the sampled data points, with one polygon per data point and the data point located in the center of the polygon. This technique, also referred to as the “nearest neighbor” method, predicts the attributes of un-sampled points based on those of the nearest sampled point and is best for qualitative (nominal) data, where other interpolation methods are not applicable. Another example is the use of nearest available weather station data, in absence of other local data (Burrough and McDonnell 1998). In contrast to this discrete method, all other methods embody a model of continuous spatial change of data, which can be described by a smooth, mathematically delineated surface.

Methods that produce smooth surfaces include various approaches that may combine regression analyses and distance-based weighted averages. As explained in more detail below, a key difference among these approaches is the criteria used to weight values in relation to distance. Criteria may include simple distance relations (e.g., inverse distance methods), minimization of variance (e.g., kriging and co-kriging), minimization of curvature, and enforcement of smoothness criteria (splining).

Interpolation techniques can be "exact" or "inexact." The former term is used in the case of an interpolation method that, for an attribute at a given, unsampled point, assigns a value identical to a measured value from a sampled point. All other interpolation methods are described as "inexact." Statistics for the differences between measured and predicted values at data points are often used to assess the performance of inexact interpolators. Interpolation methods can also be described as "global" or "local." Global techniques (e.g. inverse distance weighted averaging, IDWA) fit a model through the prediction variable over all points in the study area. Typically, global techniques do not accommodate local features well and are most often used for modeling long-range variations. Local techniques, such as splining, estimate values for an un-sampled point from a specific number of neighboring points. Consequently, local anomalies can be accommodated without affecting the value of interpolation at other points on the surface (Burrough 1986). Splining, for example, can be described as deterministic with a local stochastic component (Burrough and McDonnell 1998).

The spline method can be imagined as fitting a rubber-sheeted surface through the known points using a mathematical function. These functions allow analysts to decide between smooth curves or tight straight edges between measured points. Advantages of splining functions are that they can generate sufficiently accurate surfaces from only a few sampled points and they retain small features. A disadvantage is that they may have different minimum and maximum values than the data set and the functions are sensitive to outliers due to the inclusion of the original data values at the sample points. This is true for all exact interpolators, which are commonly used in GIS, but can present more serious problems for spline since it operates best for gently varying surfaces, i.e. those having a low variance.

Inverse Distance Weighting (IDW) is based on the assumption that the nearby values contribute more to the interpolated values than distant observations. In other words, for this method the influence of a known data point is inversely related to the distance from the unknown location that is being estimated. The advantage of IDW is that it is intuitive and efficient. This interpolation works best with evenly distributed points. Similar to the spline functions, IDW is sensitive to outliers. Furthermore, unevenly distributed data clusters results in introduced errors.

Similar to IDW, kriging uses a weighting, which assigns more influence to the nearest data points in the interpolation of values for unknown locations. Kriging, however, is not deterministic but extends the proximity weighting approach of IDW to include random components where exact point location is not known by the function. Kriging depends on spatial and statistical relationships to calculate the surface. The two-step process of kriging begins with semivariance estimations and then performs the interpolation. Some advantages of this method are the incorporation of variable interdependence and the available error surface output. A disadvantage is that it requires substantially more computing and modeling time, and kriging requires more input from the user.

Co-kriging is a form of kriging that uses additional covariates, usually more intensely sampled than the prediction variable, to assist in prediction. Co-kriging is most effective when the covariate is highly correlated with the prediction variable. To apply co-kriging one needs to model the relationship between the prediction variable and a co-variable. This is done by fitting a model through the cross-variogram. Estimation of the cross-variogram is carried out similarly to estimation of the semi-variogram.

Multivariate linear regression method is based on the simple concept of linear regression and may give good results if the linear correlation among the geographical parameters (altitude, slope, distance from the coast…) and the meteo-climatic data is consistent. As an example, in some works there are evidences of good correlation of temperature with quota, slope and latitude, while there are evidences that this method gives bad results if used to correlate rainfall with quota or other geophisical parameters. Kriging or co-kriging seems to be much better for rainfall interpolation.

Finally, A relatively new method of interpolation is the application of neural networks. A feed-forward back propagation neural network was also used by Rigol et al (2001) that considers both trend and spatial associations of climatic variables. Performance of the network was comparable to that achieved with kriging, but has the advantage that guiding variables (such as terrain) do not need to be linearly related to the interpolation data.

There are different assessment methods for the evaluation of the performance of each interpolation method (Willmott 1984). This evaluation calculates error statistics on the control stations with the recorded parameters as the observed data and the interpolated parameters as the predicted values (i.e. mean absolute error (MAE), root mean square errors

(RMSE), systematic root mean square errors (RMSEs), unsystematic root mean square errors (RMSEu), etc.)

2.3.3.2 Soil parameters

The characteristics of soils play a major role in the potential exposure of surface water from runoff and drainage routes. A number of soil properties can be important in the landscape level evaluation of soil-related exposure, including soil texture, pH, hydrologic group, organic carbon content, etc. Soil attribute data that is more comprehensive or more spatially refined can be used in Step 4 to refine the parameters used at Step 3 to be more representative of the specific conditions, crops, etc. being examined. An understanding of the organization of the tabular (attribute) and spatial soil data is the focus of this section, not an exhaustive list of relevant soil properties.

Soil data sets generally have two or more distinct components. The first component contains the soil properties (at one or more levels), and the second is the spatial component to which the attribute data is linked. Using the Soil Geographical Data Base of Europe (SGDBE, 1999) as an example, soil attribute data is related to a Soil Typological Unit (STU), of which up to 10 STUs can be related to a single Soil Mapping Unit (SMU). In addition, the European Soils Database also contains soil profile information in the Soil Profile Analytical Database of Europe (SPADE, 1999). For purposes of this section, the European Soils Database will be used as a guide.

There are a number of considerations related to the link between soil attribute and spatial information that should be examined before using a soils database. The most important is an understanding of how soil attribute information at the STU level should be summarized from multiple STUs for mapping or spatial analysis at the SMU level. Since there can be up to 10 STUs per SMU, the information held in the STU is more detailed than the SMU, although it cannot be placed in any specific geographic area *within* the SMU. The SGDBE contains a percent area field (PCAREA) that defines how much of the surface area for the SMU is covered by each related STU. To use the soil attribute data spatially, several approaches can be used: apply the entire dominant STU (largest percent area) attributes to the SMU, summarize STUs on specific individual attributes, a weighting approach used on continuous attributes, and characterization of soil attributes with other data sets at the STU level before aggregating to the SMU level.

The first method of applying the entire dominant STU attributes to the SMU has the advantage of simplicity, although overall it has the greatest amount of error. Of the 1650 SMUs, less than 10% have just a single STU, and ~450 have two or less. Therefore, the dominant STU according to SMU area may not provide an adequate representation of the soil attributes within that SMU.

The second method uses the concept that the STUs are separated based on a *number* of attributes. Therefore, when a single attribute is examined, it is likely that more than one STU will have the same value for any single attribute. To quantify this, the percent of the SMU area covered by each of the selected attributes can be summarized, calculating the SMU area with a single value. This process must be done for each attribute to be examined, but results in a greater confidence that the attribute used to characterize the entire SMU polygon represents the dominant attribute (not necessarily the dominant STU). As an example, when the surface texture field (TEXT1) is examined in this method, the average of the percent of SMU covered by a single texture is 86%. This compares to an average of 61% when only the single dominant STU texture was used. While the actual texture eventually used in both methods may be the same texture, confidence in the summarized method is greater. This method requires additional processing for both the summarization process, as well as processing for each attribute of interest.

The third method of linking STU attribute data to SMU polygons uses an area-weighted approach for ‘continuous’ attributes. ‘Continuous’ in this sense describes attributes that have a range of numeric values (e.g. soil pH) rather than a finite set of discrete classes (e.g., texture). For continuous attributes, a standard area-weighting approach can be implemented as (using pH as an example):

$$\text{pH of SMU} = (\text{PCAREA}_{STU1} * \text{pH}) + (\text{PCAREA}_{STU2} * \text{ph}) + \ldots + (\text{PCAREA}_{STUn} * \text{pH})$$

Where: PCAREA is the percent of the SMU covered by that individual STU

Since this method applies an arithmetic approach to two STU attributes (PCAREA and pH), it does not introduce any error beyond that in the original data, but it can only be applied (in this manner) for continuous attributes.

Once soil attributes are associated with the SMU polygons, they can be combined with other geo-referenced data sets (land use, weather, etc) for spatial operations. Whatever method of associating attribute data to spatial soil polygons, it should be made clear in the documentation of the process, and also on any final maps/results created using the attributed

SMUs. If possible, the errors associated with the chosen method (if any) should be expressed.

A final method performs the spatial operations combining geo-referenced environmental data sets to the SMUs first, and then examines the soils attribute data at the STU level. For example, STUs can be attributed with precipitation and temperature information. This information is then associated with each STU, and combined with soil properties (texture, land use, etc.) to compute vulnerability metrics for each STU. This method has the advantage of maintaining the detailed level of soil attributes further along in the analysis process, but makes the mapping and spatial display of results more difficult, as individual STUs cannot be mapped. Classification of SMUs can be performed on the basis of the computed metrics, and soils can then be mapped as a percentage of the SMU satisfying certain conditions.

The Soil Geographical Data Base of Europe also contains more detailed soil profile information in the Soil Profile Analytical Database of Europe. These profiles are not geo-referenced, but in some cases the profile information is explicitly linked to a specific SMU or STU. When no explicit link is given, an implicit link to STUs can be derived using soil name, and dominant/secondary soil texture.

2.3.3.3 Catchment area and characteristics

Catchments are commonly desirable as a unit of analysis for surface water, specifically for drift and surface runoff issues, since they can be used to spatially relate land area, and other characteristics, to surface water. The catchment area, cropping patterns and treated amount are parameters used in the Step 3 scenarios. Refinement of these parameters is possible using landscape-level data. Larger catchments can also be used as a unit of analysis for relative vulnerability studies in which large areas are examined to determine a suitable detailed area of study that can be ranked and/or quantified.

At the local level, catchments may sometimes be obtained through national mapping agencies or other environmental agencies. In other cases, they may need to be generated using topographic information; either hardcopy maps or appropriate digital data (hydrology, elevation, etc). It should be noted that groundwater catchments seldom follow the topographical catchments, and that this influences the final water balance of catchments. The smaller the catchment, the more the difference between topographic and groundwater catchment is likely to influence the water balance of the area being studied.

Local level catchments can be characterized using approaches presented in this chapter to refine the overall catchment area, cropping parameters, predominant soil and slope characteristics, and typical hydrologic characteristics.

At an EU-wide or national level, the size of the available catchments will not relate directly to edge-of-field scenarios in Step 3, but can be used in the site section process to create a ranking of relative vulnerability so that an appropriate (and quantifiable) detailed area can be selected for further analyses. Recently, a European-wide set of catchments has been made available in the form of the Catchment Characterization and Modelling data set (from the Agri-Environment Action managed by the Soil and Waste Unit of the JRC Institute for Environment and Sustainability) (CCM, 2003). These catchments, along with river segments, were derived primarily from 250m elevation data, and represent the best available catchment layer that spans Europe. The catchments are also classified using the Strahler ordering system for the river segment contained in the catchment, allowing for an appropriate selection of scale for each application.

2.3.3.4 Elevation & derived metrics

Elevation and derived metrics such as slope, aspect, flow accumulation, and compound topographic index (CTI) can be useful in the estimation of surface water and pesticide runoff and also provide useful landscape characteristics such as the potential presence of flowing surface water (can be used in the absence of available vector hydrography), surface water flow direction, and drainage area. Refinements in slope and drainage area have direct implications with respect to Step 3 model parameters (at the local level), while broader use of coarser-elevation derived metrics provide inputs for relative ranking of potentially vulnerable areas across the EU for Annex 1 investigations. Note that the concepts described here are strictly related to topography and therefore the underlying assumption is that exposure in the catchment is related to surface flow.

<u>Elevation</u>

Digital elevation data (DEM) can be obtained in a variety of resolutions, the usability of which must be carefully considered. For medium to fine resolution studies, a minimum post-spacing of 10 meters is suggested to capture the variability in moderate to steep relief topography for accurate analysis (Zhang and Montgomery, 1994). However, in lowlands typical of agriculturally intense areas, coarser resolution elevation data may suffice. The coarser the source elevation data, however the smaller the features that will be lost in the derived metrics that may be useful in a landscape analyses.

The use/availability of elevation data at a European-wide level vary from 1-km (GTOPO30, 1996), 250m (CCM, 2003) and 90m (SRTM, 2004). Finer resolution data must be obtained at the Member State or local level. See Section 2.4 Data Layers for Integrated Spatial Analyses for more information.

Slope

Slope can be used to determine flow direction, to generate a compound topographic index (discussed below), or used in the estimation of runoff. Slope can be generated from a DEM in several ways. There is a function in a GIS that determines the shortest and steepest direction from every cell and calculates slope in that direction. If direction-specific slopes are desired, for example the slope from crop to water, specific groups of cells may be chosen for analysis in either a vector or raster environment.

Flow Direction

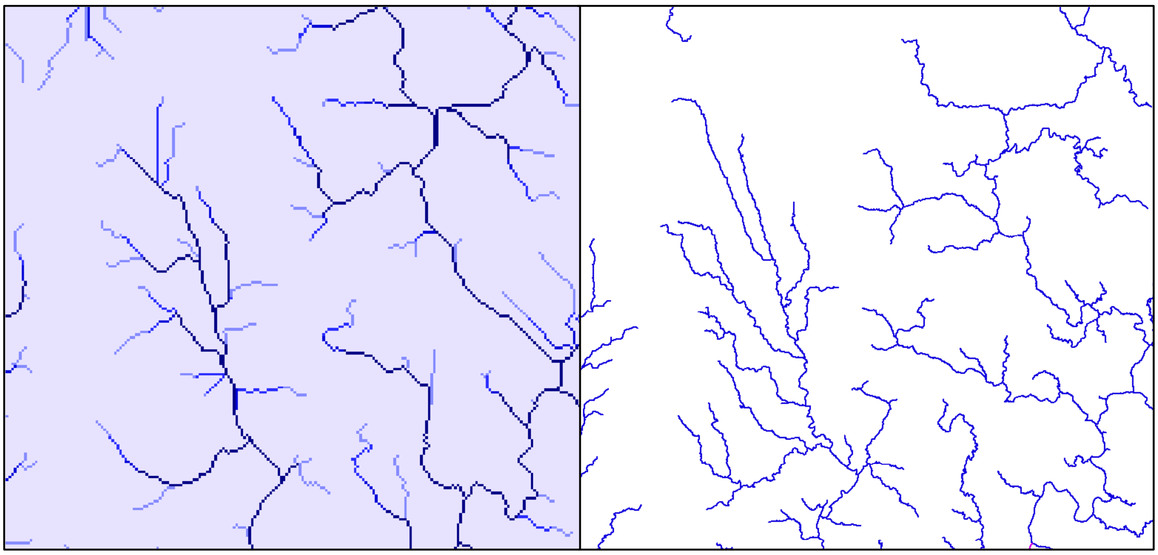
Flow direction, identified by the direction of the steepest descent, is generated from elevation data and used to generate the secondary data layers of flow accumulation and compound topographic index.

Flow Accumulation and Contributing Area

Flow accumulation is defined as the number of cells in a raster grid upslope of a target cell and is generated from the flow direction grid. Flow accumulation can be used to estimate drainage area and to identify cells with the potential to carry flowing water. Grid cells with higher flow accumulation (FA) values are those that have a greater number of upslope cells draining to them, and when contiguous, identify the path that surface water will tend to take given local topography (see Figure 2.3-3). Using flow accumulation to identify surface water may be useful in areas where there is no large-scale hydrography available, or to identify surface water consistently over large areas where constant scale/source hydrology may not be available. To accomplish this, a threshold of flow accumulation must be used to identify the path of potential water. The variation of the contributing area threshold was implemented in the CCM River and Catchment Database using 5 different factors to create a landscape stratification approach (annual rainfall, local relief, mean vegetation cover, soil transmissivity, and bedrock erodibility). (CCM, 2003)

It should be noted that the resolution of the source input elevation data, and the method used to derived potential surface water from elevation, have a significant impact on the quality of the derived potential surface water.

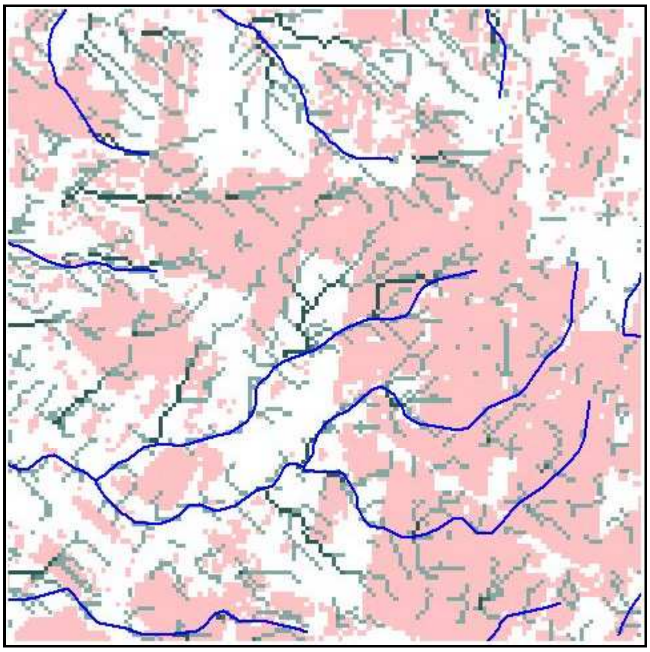
Figure 2.3-1: 1-km flow accumulation values and 1:250,000 vector hydrology



Flow Accumulation Under Crop Metric

Flow accumulation, along with crop information, can also be used to identify areas of potential contribution of pesticides to surface runoff. Lower thresholds of flow accumulation values, although not necessarily identifying perennial flow, can identify concentrations of flow during rainfall events and therefore can identify areas of potential contribution of pesticide runoff from cropped areas. Note that the method described here is strictly related to topography and therefore the underlying assumption is that exposure in the catchment is related to surface flow.

Figure 2.3-2: Flow accumulation (FA) and crop can identify potential areas of pesticide contribution to surface runoff



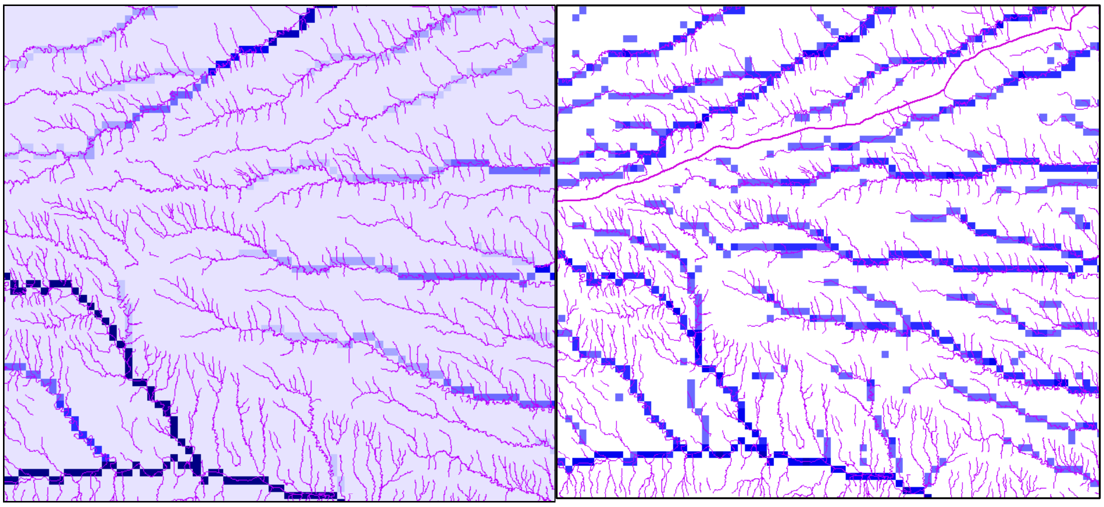
Compound Topographic Index (CTI)

The Compound Topographic Index (CTI), also known as the wetness index, is available at 1-km post-spacing from the USGS (GTOPO30, 1996) or can be calculated in a GIS. The CTI is another, slightly more complex way to identify potential flowing surface water, taking into account slope and also compressing the FA data range. The CTI is generated using the following equation:

$$CTI = \ln (FA / \tan (slope))$$

The index allows for slightly smaller channels to be identified compared to the flow accumulation metric (see Figure 2.3-3) but can also result in greater commission error (cells that identify flowing water where there is none). The commission and omission errors can be somewhat mitigated by defining criteria that take into account neighboring cell values and local value ranges. For example, if the difference between the mean and the maximum local values in very small, the maximum CTI value is less likely to identify truly convergent flow.

Figure 2.3-3: 1000-m DEM generated FA (left) and CTI (right) and 1:100.000 vector hydrology.



2.3.3.5 Hydrologic drainage density

The hydrologic drainage density of an agricultural area may be an indicator of relative potential exposure to surface water from agricultural pesticides. Drainage density is calculated as an indicator of the average drainage capacity of the hydrographic network over a certain area. Drainage density can be examined using a grid-cell approach, catchment level, or by using a “moving window”. In addition, refined knowledge of the hydrologic function of the surface water can be used to modify the drainage density used for specific purposes.

The relative ranking of hydrologic drainage density can be used to compare regions (both within a single MS and between MSs, or perhaps in the development of new scenarios), and can also be an important part of the broad site selection process, in order to ensure that the final area studied represents the level of agricultural / hydrological interaction desired.

It should be noted that drainage density metrics are based on the presence of surface water in a GIS data set, and hence do not include the presence of artificial drainage (such as pipe drains, mole drains, very small ditches, etc) that are also a component in movement of potentially exposed water. In addition, in some areas of the EU (e.g. Denmark), some natural streams are “piped” (i.e., not open to the surface), and should not be considered as an “open” stream for the purposes of determining potential vulnerability.

Drainage density was quantified in the Hydrologic Atlas of Germany addressing issues relating to inclusion of flowing and static surface water, unit of analysis and scale (HAD, 2000). Input data sets of hydrology were between scales of 1:200.000 (for flowing) and 1:1.000.000 (for static water) for final production of 1:2.000.000 scale paper maps (A2 size).

The total length of flowing water was added to the total perimeter length of static water to produce a single drainage density value per unit area (in km/km^2). This method is an attempt to adjust for areas of low flowing water density, but may have a high density of surface water when both rivers and lakes are combined.

Approaches used to length density per area based on 'natural areal units' and grid cells were both discarded in favor of using a circular area around a central grid cell. 'Natural areal units' having similar environmental properties (soil; geology, land cover, etc.) were desired, but available data was not sufficient at that time. The grid cell approach was not used because at the scale of input data sets and output map, small grid cells (<5 km) produced spatially scattered results, and larger grid cell sizes tended to lose boundaries of regional characteristics.

To capture regional characteristics and also identify areas where sharp local changes occur, a moving circular window based on a grid cell was chosen. A window of 75 km^2 was used in which the total length of flowing and static water was summed. The moving window was applied to each grid cell in turn, so that the total surface water length for each cell was composed of some of the same surface water as neighboring cells, thus giving some "smoothness" to the output map (1:2.000.000 scale), while retaining some local variation.

Comparison of results with those generated from samples of larger scale data sets (1:25.000 to 1:100.000) showed strong correlations, but the strength of the correlations varied with different areas. While relevant to large areas such as an atlas for all of Germany, this combination of method and data may not be appropriate to study much smaller areas.

Drainage density at a more local scale within a region was quantified in northern Italy using a grid call approach (Verro et al., 2002). In order to map drainage density with a spatially uniform distribution the calculation is referred to a regular grid of squared cells. The resolution (i.e. dimension of the squared cell) should be defined in accordance with the desired scale of analysis and with the resolution of the others layers involved in the modeling procedure.

For each cell, an index of drainage density was obtained by intersecting all the hydrographic layers with the cell's boundaries, then summing the total length of the ditches and of the secondary natural tributaries resulting from the intersection. The ratio between the total length and the area of the cell represents the density of the hydrographic network. The different role of each natural or artificial tributary is then considered in order to weight the relative importance for the drainage system and to obtain the final drainage density distribution. For this study, the hydrology was divided into two classes: irrigation ditches (artificial channels for water distribution to cultivated lands that don't usually generate inflow into natural water bodies) and drainage ditches (mainly natural secondary tributaries that may generate inflow into natural water bodies).

The function of the ditch (irrigation or drainage) was also taken into account, and was related to it's geographical location. (i.e. In Lombardia some representative sub areas were studied and it resulted that a separation of ditch function exists across the "spring line", defined as the line where the piezometric surface reaches the topographic surface. Above this line 10% of ditches play a drainage function while downstream this percentage increases to 50%). According to the ditch function for each grid cell (irrigation or drainage according to location) only a percentage of the total length of the contained ditches may be used in the calculation.

This study required GIS layers of the ditches and secondary natural tributaries (the scale should be chosen according to the other layers), and any eventual physical variable useful to distinguish the function of the ditches, if there is a distinction of the ditch function.

The regular grid was built as a vector layer of squared polygons with equal area, defined by the user and representing the spatial resolution of all the process. All the ditches, channels, natural tributaries need to exist as vector layers of lines. All the layers representing the water lines were clipped by the cells grid in order to exactly calculate the length of ditches inside each cell. For each cell, a union of the clipped ditches was performed and the total length is summarized. For each cell, the total length is reduced considering the eventual ditch function (based on the cell's location). For each cell, the ratio between the modified total length of ditches and the cell area was calculated.

Drainage density index (DDI) for each grid cell:

DDI (dimensionless) = L (m) x W(m)/A (m)

Where: **L** is the total length of drainage ditches in the considered cell

W is the width of drainage ditches (An average W of 1 m can be assumed for every ditch)

A is the surface area of the grid cell.

A linear relationship is assumed to use the output of this calculation to calculate the percentage of applied AI lost by drift (% D) in function of the drainage density:

$$\% D = k \times DDI \times 100$$

Where: **k** is a proportionality factor depending on crop and crop stage according to %drift values reported by Ganzelmeier, and distances between sources (in meters).

2.3.3.6 Cropping density

Estimation of cropping density is a common modeling parameter to refine as it is relatively straightforward to estimate and provides an understandable and transparent modification to the Step 3 scenarios. There are a number of methods that can be employed to derive cropping density, depending on the unit of analysis, available data and spatial processing complexity.

It should be noted that all approaches to determining cropping density (and other factors involving crop information) relate to information about crop location/area for a single point in time (e.g., when remotely sensed imagery was acquired, or when the crop statistics were gathered). Certain land uses change in time more rapidly than others (i.e., permanent crops like vines & orchards vs. annual crops such as maize and sunflower). The specific crops and issues being addresses should guide the user as to the most appropriate data and method to use, as well as how to interpret the results. For example, cropping density for vines based on 3-year old land cover has more confidence than cropping density for maize based on 3-year-old data. Likewise, cropping density for a broader crop class, such as arable crops, will also have more confidence than cropping density for an individual crop such as maize. The Step 3 scenarios assume various densities of cropping, and these assumptions can be refined using spatial and tabular data for a given geographic region.

Because surface water bodies are the primary concern, one method of computing cropping density utilizes “margins” around individual water bodies. These margins are a notional area surrounding surface water out to a specific distance, used to characterize the area immediately surrounding the target. Specific distances from the water body define the margin (for example, 20 metres, 100 metres, 200 metres, etc). In this manner, the area most likely to impact the water body is characterized in terms of cropping density. For concerns about drift, the margin distance may only be 50 metres, where runoff and drainage may have much larger distances. The general idea of this approach is to spatially target the area most likely to adversely impact the water body. This approach may be difficult to implement over a large area, as each water body must be processed individually, requiring some level of automation within the GIS.

In some cases, more likely with runoff and drainage issues, a catchment level analysis may be more appropriate, as all the area within the catchment has the potential to impact the water body. In this case, the cropping density of each catchment can be calculated and used for higher tier modeling. The level of catchment will have an impact on the implementation of this method, with smaller catchments yielding more location-specific results, but requiring more processing and a greater level of detail required for the catchment data.

A gridded approach (e.g., using 5 x 5 km areas to summarize cropping density) has the advantages of not requiring as much GIS processing and automation, and also does not require detailed spatial data sets, as a grid cell (artificially delineated) is the unit of analysis, not a water body or catchment that must be defined from another source. The size of the grid cells should be adjusted to suit the needs of the analysis, provided it falls within the suitable application of the other data sets (land cover data set).

In all cases, regardless of the unit of analysis, the percent crop landscape factor is calculated as the *crop area / total area*. The *crop area* and the *total area* are automatically calculated by the GIS using a standard spatial intersection operation, resulting in polygons/grid cells with attributes of both the land cover (crop) and unit of analysis (margin, catchment, or grid cell (e.g., 5 x 5 km area)).

One issue to consider when calculating crop density is the source of the crop area measurements. This can be derived solely from the land cover data (which identifies the spatial location of cropped area), from crop statistics (usually associated with some level of administrative unit), or both. If only land cover data are used, it is important to remember that the percent crop values generated will be affected by the quality and method data collection and generation of the land cover data.   For example, the Minimum Mapping Unit (MMU) size of the CORINE Land Cover data (CORINE, 2000) will affect the total area of crop using the spatial data sets. The MMU specifies that areas of less than 25 hectares are not represented as individual polygons, and will either be combined with other areas into a "mixed" class, or not represented at all. Therefore, calculating the total area of crop land using only the CORINE Land Cover data will have different results than other spatial data sets, and also be different from crop statistics.

When more accurate crop area measurements are required (for example, when combined with application rates and local monitoring data), crop statistics are often employed. These statistics are usually delineated using administrative boundaries (such as NUTS levels). Administrative boundaries typically do not coincide with catchments or other hydrologically relevant delineations, so some method of interpolation is needed to calculate useful cropping densities for a catchment (or grid cell). This is commonly done using an area-weighted approach, where the proportion of crop area (up to 100%) from the crop statistics are allocated to each catchment based on the proportion of total administrative unit area within the catchment. For example, if 40% of an administrative unit is within a catchment, that catchment is allocated 40% of the crop area from the crop statistics. For each catchment, the total amount of crop (from all contributing administrative units) are summed to determine the total amount of crop in the catchment. This method can also be applied to the grid-cell approach.

One disadvantage of the area-weighted approach is that is assumes an even distribution of crop within the administrative unit. Depending on the homogeneity of land cover and size of the administrative unit, this assumption may not be acceptable. To address this, an approach that combines the use of land cover data and crop statistics may be used by performing a spatial operation in the GIS which combines the land cover and administrative unit data sets. This approach uses the spatial land cover data to identify only those locations that comprise crop within the administrative unit, generating a more spatially refined value for the location for each catchment / administrative unit combination. The crop statistics are then used to allocate the total amount of crop to each catchment area, similar to the area-weighted approach. This method utilizes the strengths of both the spatial crop locations, and the 'official' crop statistics.

This method still assumes an even distribution of crop across the crop locations based on the spatial data. The confidence in this assumption depends on the classification of the spatial data and the crop(s) being examined. For example, if the spatial data simply has a single "crop land" category, the assumption that arable crops are evenly distributed over the spatial data has more confidence than if a single crop, such as maize, is being examined.

2.3.3.7 Buffer width

Another primary factor in assessing potential exposure to surface water is width (and composition) of naturally occurring buffers between cropped area and surface water. These buffer areas may supply some natural mitigation to spray drift or the runoff process. In order to quantify the buffer width and composition, it is important to define the spatial component of the buffer. Commonly, this is defined as the area located between crop and water. The width of the naturally occurring buffer is a direct parameter in the Step 3 models, and provides an easily understood method for modifying Step 3 scenarios in the Step 4 process.

While much of the discussion regarding buffers is related to spray drift, it should be noted that buffers are an important factor in mitigation of runoff as well (see mitigation sections in Volume 1 and Volume 2 of this report). While the concept of a buffer is equally valid for both drift and runoff, the characteristics of the buffer that promote mitigation are different for the two processes. For spray drift, the width, height and density of the buffer are the primary factors (see sections on mitigation of spray drift), therefore a continuous buffer of hedgerows would be ideal. For runoff, width is still important, but the height of the buffer vegetation is less important, and other factors such as slope and soils properties in the buffer will have more importance. Therefore, a grassy buffer on a relatively flat area would be more desirable.

Note that in cases of direct adjacency of crop to water, no buffer exists. It is important to decide, and make clear in presentation of results, whether cases of direct adjacency will be used in calculating the average buffer width results.

Due to the fact that buffers tend to be relatively small, as small as several metres, detailed analysis requires spatial data of sufficient scale / resolution to identify and quantify the buffers in the landscape. This may include both aerial (~1.0 metre) and satellite imagery (10 – 30 metres) and scales from 1:10.000 to 1:50.000.

Because the presence and width of a buffer varies along the perimeter of a water body, a detailed approach quantifies the buffer as it varies, and computes a generalized buffer width for the water body. For example, using all the measurements of buffer width to compute an average buffer width for the water body. In this case, each water body has a specific buffer width. More generalized buffer widths at the catchment or grid cell level may also be generated using less detailed measurements in conjunction with other knowledge geographic regions.

In the case of individual water body buffer measurements, two approaches have been successfully implemented. The first uses the presence of crop within a specific distance of water, and determines the shortest distance to the water body for set points along the perimeter of the crop field (polygon) (Kay, 1997). The second uses sampling point placed along the perimeter of the water body to determine the distance and direction in which crop is located (Holmes, 2002). This method allows for a convenient way in which to characterize the perimeter of the water body as percent directly adjacent to crop, percent buffered (out to *x* metres) and percent non-cropped (out to *x* metres). See following section on perimeter composition for more information.

One implementation of the first method places sample points along the perimeter of crop fields / polygons and, using the GIS, “draws” a line to the closest point on the water body perimeter. These “transects” then represent the width of the buffer between crop (at that point) and the water body. Sample points that are on the far side of fields (from the water body) are discarded from the analysis. For each water body, the complete set of transects can then be examined to compute an average buffer width for a single water body.

One drawback to this approach is that the structure of the water body (sinuosity, etc) has a large impact on the determination of the shortest distance to water. For example, a field near a bend in the stream may have a disproportionate number of transects drawn to the corner (since it is the shortest distance), when in fact a more common sense approach might distribute the spacing of the transects more evenly, in order to get a less extreme measurement of buffer width. Also, the transects created during this process can be used to quantify the buffer area (essentially make a polygon of all the transects), and may not produce a buffer of sufficient area. This method is best used when the most conservative buffer width measurements are desired, to the detriment of other metrics such as buffer composition and water body perimeter composition.

The second method that utilizes sample points along the perimeter of the water body to determine the distance and direction in which crop is located provides a balance between the buffer width, buffer composition and perimeter composition metrics. This method uses multiple directions from evenly spaced sample points (usually 8 directions) to determine distance to crop. Therefore, the shortest distance to crop for each point can be used to calculate buffer width, while the even distribution of sample points along the perimeter of the water body (not along the crop perimeter) results in the ability to generate meaningful exposure metrics about the water body perimeter. (Alternatively, lines from each sample point can be drawn outward from the water body, in an orthogonal direction to the flow path at each sample point. This has the advantage of fewer lines, but presents a new set of issues related to the instances where the orthogonal line on a bend in a stream does not yield the desired results. In addition, the 8-direction approach generates data which can be used later for a directional analysis of the landscape.)

Implementation of this method can be CPU and data intensive in the GIS, but produces a robust set of spatial and tabular data for generation of metrics. This method can be summarized as:

- Place sample points along the perimeter of each water body at a specified interval
- Extend lines from each point out in eight (or four) directions to a specified distance (depending on the maximum influence of the particular crop type)
- Intersect these lines with the land cover classification
- Process the resulting information on water body, sample point, direction, and land cover to determine distances to crop and intervening land cover (used in buffer composition analyses)
- Calculate the buffer width for each water body

The spacing interval depends on the scale/resolution of the data, generally ranging from 5 metres for ~1.0 metre land cover data from aerial imagery, to 10 or 20 metre spacing for satellite image based land cover. Since land cover is more likely to change in shorter distances with fine resolution data, the decreased interval spacing can be used to accommodate the changing land cover. Likewise, since land cover derived from 10, 20 or 30m satellite data is unlikely to change with short distances (such as 5 metres), perimeter sample points spaced at 10 or 20m intervals are likely to be sufficient to capture the changes in the immediate land cover around a water body.

Lines are extended from each sample point to a distance that sufficiently characterizes the potential for crop within that distance to impact the water body. For example, the distance for potential drift impact from arable crops would likely be less than the distance used for orchards or vines. The lines are extended out from the water body in eight directions (N, NE, E, SE, S, SW, W, and NW) to adequately capture the potential impact of different directions. Fewer or more directions could be implemented to either reduce processing and CPU time (fewer directions) or to increase the density of coverage for the surrounding landscape (more directions).

The complete set of lines radiating from the perimeter sample points are intersected with the land cover classification using the GIS. The results then give the length of each type of land cover passed through from water body to crop location, for each water body, point and direction. Depending on the number of water bodies, perimeter sample point spacing, length of lines created, and the scale/resolution of the land cover data; the amount of data generated can be very large, several millions of individual records.

The information on length of each type of land cover passed through from water body to crop must be pre-processed, analyzed and summarized to produce the ‘distance to crop’ measurement needed for characterize the buffer width landscape factor. Each line extending from a sample point in a given direction is composed of one or more segments, depending on how many land cover polygons it passed through. These segments must be ordered so that they can be examined in the correct sequence, from water body to outwards. Once ordered, each segment can be examined in turn, to determine if the land cover of that segment is crop. If not, the length of that segment is added to a running total until crop is found. Once crop is found, the sum of the previous segment lengths is recorded. This process is repeated for each of the directions for each sample point.

Given the distance to crop for each sample point and direction. The buffer width for the water body can be computed using the direction containing the shortest distance to crop for each sample point, and computing the average of these values for the water body.

This method also produces conservative buffer width measurements (by using the shortest distance to crop based on 8 directions), and also allows for easy computation of buffer composition and water body perimeter composition using the same set of data (see following sections).

2.3.3.8 Buffer composition

The composition of naturally occurring buffers between cropped area and surface water is an important factor in understanding the potential exposure of surface water. Using spatial data sets comprising surface water and crop locations, the buffer area (that area located between crop and water) can be quantified in a GIS. In addition to the buffer width (described above), the relevant characteristics of the buffer can be described. This can include the relative amounts of different land covers in the buffer, soil properties, slope, etc. The potential mitigating effect of these characteristics can be used to refine the potential exposure of neighboring water bodies. While the concept of a buffer is equally valid for both drift and runoff, the characteristics of the buffer that promote mitigation are different for the two processes.

As with buffer width, there are several methods of quantifying the buffer composition. One method examines the buffer area as a polygon, which is bounded on two sides by water and crop, and on the other sides by some method of determination (i.e., shortest distance to crop from edges of crop fields). An alternative method utilizes water body perimeter sampling points and lines extending from these points (as described in the buffer width section) to quantify the different land covers these lines pass though before encountering crop. These two methods are presented in the context of quantifying the amounts of various land covers with the buffers. Once the buffer is defined spatially, other characteristics (such as soils, slope, etc) can also be quantified.

To create a polygon of the buffer area, some method of determining the lateral extent of the buffer is needed. Two sides of the buffer are already bounded by the water body and the crop field. The other sides need to be delineated in some manner. One method uses points placed on the crop perimeter, snapping a line to the closest point on the water body perimeter. These "transect" lines can be used to measure buffer width, but the two extreme lateral transects can also be used, in conjunction with the water body and crop polygon, to create a buffer polygon. While these transects characterize the most conservative buffer widths, they may not always characterize the buffer *area* properly. For example, a field near a bend in the stream may have the extreme lateral transects drawn to the corner (since it is the shortest distance), when a more common sense approach might draw lines more orthogonal to the water body to define the buffer area. An approach to extend lines outward orthogonally from the water body may capture a more generalized buffer area. This approach may have difficulties for implementation, due to the sinuous nature of flowing water bodies and the generation and relevance of an orthogonal direction at any single point along the perimeter.

A second approach to quantifying the buffer is to use a set of lines extending outward from sample points along the perimeter of the water body (see buffer width section for methodology). Since these lines extend outward from the perimeter in multiple directions (eight directions), this entire set of crisscrossing lines intersected with the land cover provides an adequate representation of the buffer area. These lines are a linear representation of an area measurement (the buffer polygon) in multiple directions. The total linear distance of each land cover encountered prior to crop can be summarized and expressed as a percentage of the total linear distance between crop and water, and therefore represents the percent of the buffer area composed of individual land cover types. Because this approach uses the same spatial data set as the buffer width (and also crop proximity and perimeter composition described below), it is an efficient method to create all four landscape factors.

2.3.3.9 Water body perimeter composition

The composition of the water body perimeter describes the relative proportions of perimeter that are in different exposure categories (directly adjacent, buffered, and non-cropped). This may be important because the spatial relationship of a water body to crop (the potential exposure) varies along the length / perimeter. This measurement provides an indication of the variability of potential exposure along the water body perimeter, beyond simply a single PECsw value. While this is not a standard parameter in Step 3 models, knowledge of this exposure variability can enhance the understanding of the influence of the landscape related to exposure.

One method of categorizing the perimeter according to potential exposure uses three categories: crop directly adjacent to the water body, crop within a specific distance of the water body (buffered), and no crop within a specific distance of the water body (non-cropped). Using these categories, it is possible to quantify how much of the water body perimeter is subject to the greatest exposure (direct adjacency), how much is subject to exposure from crop out to a specific distance (buffered, and subsequently the average buffer width), and how much is likely to be unaffected by crop out to a specific distance (non-cropped).

To quantify the perimeter composition into one of three categories, it is possible to use the same set of lines extending outward from sample points along the perimeter of the water body as implemented in the buffer width/composition sections (see buffer width section for methodology). The shortest distance to crop (of the 8 directions) is examined for each sample point and the point is classified as: directly adjacent if the shortest distance to crop is less than 1 metre, buffered if the shortest distance to crop is one metre or more but less than the maximum distance (as defined by the study), and non-cropped is the shortest distance to crop from all 8 directions is greater than the specified maximum. The maximum distance should be one that sufficiently characterizes the potential for crop within that distance to impact the water body. For example, the distance for potential drift impact from arable crops would likely be less than the distance used for orchards or vines. Once each sample point is categorized, the percent of the perimeter in each category can be estimated as the percent of total points in that category. This assumes that the same points sufficiently characterize the potential for land cover to change within that distance.

2.3.3.10 Crop proximity to surface water

Measurements of buffer width give an indication of the distance between crop and water, when crop is located within a specific distance. Perimeter composition compliments the buffer width information by quantifying the extent to which the water body is buffered. The crop proximity metric combines both the measured distances to crop along the perimeter, with the percentages of water body perimeter that fall within specific classes, into a single index. Therefore, the crop proximity metric provides a single value that expresses extent of potential exposure (as % of perimeter exposed) and the intensity of the exposure (as distance from crop to water). While this is not a standard parameter in Step 3 models, knowledge of this exposure variability can enhance the understanding of the influence of the landscape related to exposure.

For example, consider the case where buffer width and perimeter composition measurements have been calculated according to the preceding sections using sample points placed on the water body perimeter. In this example, assume lines were extended from the water body out to 100 metres to determine if crop is proximate to the water body. The perimeter composition reports the percent of the perimeter that is directly adjacent (<1 metre to crop), buffered (1 to 100 metres to crop), and non-cropped (more than 100 metres to crop). The buffer width results report the average distance between water and crop (out to 100m), not including cases of direct adjacency. The single crop proximity metric is a value between 0 (100% directly adjacent perimeter) and 100 (100% non-cropped perimeter) and can be calculated as:

Proximity = (% adjacent * 0) + (% buffered * avg_buffer_width) + (% non-cropped * 100)

The resulting metric will be scaled between 0 and 100, since direct adjacency is represented by a value of 0, the maximum buffer width is 100 metres, and non-cropped perimeter is represented by a value of 100. In this case, the crop proximity metric can be loosely equated to a generalized ‘distance to crop’ metric (out to 100 metres) for the water body as a whole.

Another approach to quantifying the proximity of cropped fields to surface water uses a field-based unit of analysis and a raster GIS (Gutsche, 2002). In this approach, both land cover and surface water are in a raster format, with 5-metre grid cells. The source of the land cover data is the Authoritative Topographic Cartographic Information System DLM/25 (ATKIS). The ATKIS land cover is originally in vector format, with each feature (polygon) having a unique identifier, including crop fields. During the vector to raster conversion process, the field identifier was maintained, and allowed fields to be examined as a discrete unit.

To quantify the proximity of the field to surface water, the Euclidian distance between each field cell (5x5 metres) to the nearest section of a water body is calculated. Each cell associated with the field then has a shortest distance to water. The smallest of these can be used as a conservative metric for the proximity of the field to surface water, or some arithmetic combination of the field pixels, such as average or percentile, could also be used to calculate a single proximity value for the field.

2.3.3.11 Wind speed & direction

In order to estimate pesticide's drift effect on surface water ecosystems, drift phenomena may be modelled or simulated at the local level. It would be misleading to calculate drift trajectories considering only pesticides' application techniques, buffers or physical barriers, without including wind effects.

The drift component in the current FOCUS approach uses drift functions derived from experimental data (BBA, 2000). The BBA 2000 spray drift data were generated under specific conditions, which need to be considered to better understand the impact on drift estimation and Step 4 approaches. For each crop type, a number of trials were conducted to determine off-crop drift. To estimate off-crop drift, the downwind edge of the field was sprayed with tracer. Drift deposits at ground level were sampled from downwind areas free of vegetation at a variety of distances from the crop edge. Data obtained in the study were subject to a statistical analysis to examine the distribution of drift values. A 90th percentile value was calculated from measured drift values at each distance to estimate the worst-case drift value. Regional wind speeds and variations in vegetation downwind may be examined, resulting in potentially different drift values (larger or smaller depending on the factor).

In addition, the current assumption at Step 3 is that the wind always blows toward the water body, resulting in the shortest path possible for drift deposition. Variations in crop distance and direction across a landscape can be examined to illustrate the variation of exposure to a water body based on direction. Water bodies can be evaluated for drift input based on individual wind directions, even to the level in which the "realistic worst case" distance and direction combination for each water body can be determined and reported on.

Information on wind speed and the local effect of direction can be estimated using existing data sets, or local information can be collected. If existing data are to be used, variations in speed and direction can be implemented (after providing justification for the reason) and the subsequent impact on exposure (i.e., drift deposition) can be examined. Section 2.3.4.1 presents several approaches to estimating spray drift deposition, in which the factors of wind speed and direction can be implemented. If alternate drift deposition rates can be supported due to variations in wind speed, the resulting drift rates can be incorporated into the landscape exposure analysis (as described in Section 2.3.4.1) by providing alternate drift curves. Drift deposition can also be esimtated at the landscape level for individual wind directions for each water body (or sampling point along each water body). In this manner, a direction-dependent drift PEC can be computed for each water body, so that dirft deposition is only occurring from one direction for a single PEC estimation. In the detailed analysis of drift, an added step of conservatism could be added, so that drift for each sampling point along the length of the water body is computed independently using the direction that contains the shortest distance to crop for that point. Further descriptions of drift estimation using landscape level data are presented in Section 2.3.4.1.

Local information on wind speed and direction may also be collected and elaborated in order to evaluate wind effect on drift phenomena and different approaches may be used, on different spatial and temporal scales, depending on the availability of data:

- For long term averaged scenarios (seasonal or annual), the seasonal or annual averages of wind speed associated to their most frequent directions (wind-roses) may be considered in order to add to the general scenario a “wind contribution factor”. This could be acceptable for screening exercises or for evaluations on wide spatial domains.
- Also for more accurate modelling activities, in absence of any measurements or monitoring data on-site there are no alternatives than using general averages (prevailing winds) extrapolating wind speed and direction from the closest location where there are measurement point that could be considered representative also if not placed locally and considering this as an “average situation”.
- Another possibility is the extrapolation of ground level wind speed and direction starting from the general geostrophic wind field (obtained from the local or regional Numerical Weather Prediction). With rather simple algorithms the vertical wind profile can be reproduced according with Eckman theory; some

parameterization is necessary (horizontal wind friction velocity, Von Karman constant, height of the Planetary Boundary Layer …) but vertical wind profiles may be estimated for stable and unstable conditions, in absence of any other measured data. Interpolation is then necessary for horizontal spatialization.

- If some instrumentation is available (or can be placed) on-site to measure ground level wind speed and direction, the local wind field can be reproduced with good accuracy. Anemometers may be positioned in strategic points at some meters from the ground and once collected all the measurements, short term or long term averages can be calculated and used to reproduce the averaged wind field covering all the area of interest. (An alternative to classic anemometers is the sonic anemometer – SODAR – by which is possible to scan the whole vertical wind and temperature profile, but it is a very expansive instrument, may be too much sophisticated for the aims of pesticides drift models).

The latter case is the most suitable in order to supply a spatialized wind field to the drift models at local or at field scale, but interpolation techniques must be carefully assessed, discussed and evaluated. Different studies show that geostatistics techniques may be applied to ground level wind field spatial interpolation, with some limitations. Kriging and thin smoothed splines seem to obtain better results than other methods, especially for low spatial resolution, wide geographical areas and where orography is not too much relevant for wind field spatial variations (i.e. Engel and Michael made an attempt to evaluate the Barnes' method as a simplified solution to build wind fields. After an evaluation-comparison of Barnes' Method and Kriging for estimating the low level wind field, Barnes' method proved to be the less complicated to implement, but Kriging provided a more accurate estimate; this shows that at least a geostatistical technique should be used for wind fields spatialisation, renouncing to simpler methods). In meteorology and climatology spatialisation of continuous scalar data is of major importance but for the spatialisation of vector data (e.g. wind), a separation into two scalar data is frequently used. Then these scalar data are interpolated independently by spatialisation schemes for scalar data.

Nevertheless, wind is a three-dimensional phenomena expressed by vector direction and module, and it is a difficult element to estimate locally using simple interpolation. According to some studies (COST719) the only reliable way to this seems to be the application of a physical approach. Downtransformation or fine scale dynamical models are applied. These methods are purely based on physical laws and/or empirical parameterisations. Some of these methods involve NWP (numerical weather prediction) model output, but others work in combination or solely with observations. NWP models usually produce large-scale output, the purpose of the so-called downscaling (or downtransformation) methods is to increase the spatial resolution of the NWP output by adding information of the topography (like e.g. roughness length or albedo) on a higher resolution basis. Another solution would be to increase the resolution of the NWP model itself, however, this implies a relatively small calculation area and high computational costs. Besides, when e.g. site-specific wind speed in a heterogeneous terrain is needed the horizontal resolution has to be unrealistic high. On the other hand, disadvantage of the downscaling techniques is the vulnerability for the quality of the NWP model output. Moreover, some of the downscaling techniques can be inaccurate during very stable atmospheric conditions.

All the former considerations, together with the wide range of temporal and spatial scales involved, make it difficult to give a simplified generalization of physical modelling techniques and a simple description of physical methods to be considered valid everywhere and in whatever situation.

In this report a wind field processor is suggested as an example of physical method that can be applied everywhere, at different degrees of availability of input data. In this case the wind-field processor is a mathematical model working on a 3D gridded domain and reproducing the three-dimensional wind field. The mathematical model suggested here (WINDS – Ratto et al., 1990; Mazzino et al., 1994; Ratto et al., 1994; Ruaro et al., 1995) is just one possible model among many, but it is a useful example to show how the complex operation of building a wind field can be performed using a user-friendly informatics tool. Furthermore this model has been validated in many scientific exercises, coupled with air pollution dispersion models. Anyway there are other similar models, based on mass conservation, 3D grids and terrain-following coordinates.

WINDS (Wind-field Interpolation by Non-Divergent Schemes) is a diagnostic mass-consistent model for simulation of the three-dimensional wind field in complex terrain at the mesoscale level. WINDS was designed at the Department of Physics, University of Genova, Italy as an evolution of the AIOLOS model (Lalas, 1985; Lalas et al., 1988). Both models have the origin in the well-known NOABL model (Philips, 1979) with diverse modifications to construct more reliable “first-guess” wind profiles, introducing surface roughness and stability effects in the Planetary Boundary Layer (PBL). Dependent on the available data the code can construct the initial wind field using ground station's data and/or geostrophic wind or observed experimental vertical profiles (for example SODAR-data). Atmospheric stability and different surface roughness effects on the PBL structure are taken into account by means of theoretical expressions (Zilitinkevich, 1989). The typical spatial horizontal scale is between 2 and 200 km (mesoscale), while the vertical one is typically about 2 km . The typical model time scale is one hour, but with regard to the characteristics of the atmospheric turbulence spectral gap, it is possible to use WINDS on the base of a time scale of 15-10 minutes if measurements to initialise the code are available. Mathematically, the calculation of three-dimensional wind field is achieved by a two step procedure: first an initial guess for the wind field is constructed, and then an adjustment is made to eliminate divergence. The first step includes methods for horizontal and vertical interpolation of available wind data, or calculation of “initial” (or “first guess”) wind field. The second step includes a procedure for minimum possible modification to the “first guess” wind field so that the resulting (or “final”) wind field satisfies mass conservation. The mathematical formulation of the mass-consistent model WINDS uses a calculus-of-variations approach originally developed by Sasaki (1958, 1970). The variational problem is to minimise the variance of the difference between the adjusted and the initial wind field subject to the constraint that the divergence should vanish.

The interest in this tool is that the user can run the model initialising the program also with very few data. There is a range of seven tiers of initialisation, starting with uniform wind only (any station wind data required) or with geostrophic wind (any station required) or with ground weather station, SODAR, etc. Each different kind of initialisation offers a different level of accuracy. Orography and surface roughness may be declared constant, but if they are available (in raster format) they will allow the model to produce results with higher accuracy. The final result is the wind field in a three-dimensional grid where at each cell is calculated the wind direction and velocity. For the purposes of pesticide’s drift modelling, the third dimension of the grid domain is not as much important as the bi-dimensional field at ground level. This results in a grid with the values of wind speed and direction assigned to each cell, so it may be used directly for contouring and tracing isopleths of wind velocity. Then the grid of the wind-field can be combined with other raster data of land-use, land-cover and so on, in any calculation to be performed inside a GIS environment, introducing the wind velocity and direction as a landscape factor.

2.3.4 Exposure estimates

Following the tiered approach given with the definitions of the FOCUS surface water working group (FOCUS, 2002), a Step 4 approach of the aquatic exposure assessment usually will be preceded by performing Step1 to 3. With the Step 3 assessments fundamental experiences on the major characteristics of the predicted aquatic exposure have been achieved on the basis of the definitions of the FOCUSsw scenarios and models, *i.e.* the necessity for refinements as well as first ideas of areas for refinements (*e.g.* crucial parameters governing the dominant entry route) have been identified.

The next step (leading over to Step 4) could aim at the determination of most significant Step3 model parameters from spatial data for most relevant PPP use regions, in order to characterise the actual environmental conditions more realistically. The complexity and costs of a spatial modelling using Step3 models depend on the entry route(s) to be investigated and the model parameters to be characterised using spatial data: e.g. as known from empirical studies, spray drift depositions in the vicinity of spray applications strongly depend on the distance from the sprayed field. Therefore, this distance will play a key role in the analysis of real environmental conditions. Refined estimates for potential runoff entries into surface water bodies are more complex, as single dominant driving factors are not apparent, and moreover the potential loadings depend on the properties of the chemical. For these (and further) reasons currently a generic empirical database on runoff loadings in relation to relevant environmental conditions (e.g. soil properties, slope, weather, crop, intervening zone conditions, cultivation practice, etc) and compound properties does not exist. Nevertheless, refinements of the exposure assessments are possible as long as the remaining "unknowns in the equations/assumptions" can be reasonably estimated following the realistic worst case principle (e.g. refinements can exclusively be based on crop-water body distance and the conditions of the intervening zone using appropriate spatial data and categorised runoff reduction factors, whilst keeping the remaining definitions of the current FOCUS Step3 PRZM approach).

The spatial data applied to derive Step3 model input parameters should fit to the scale on which the aquatic risk assessment will be done (or the regulatory decisions will be made), *i.e.* the thematic, spatial and temporal accuracy has to be appropriate to base Step3 model calculations and the risk assessment upon.

Regardless how simple the basic concepts of Step3 model refinements using spatial data are, the modelling of local environmental conditions comes with many thousands of local situations to be processed, calculated, managed and evaluated, using a number of different tools (GIS, classification software, Step3 models, Statistic packages, etc) potentially including the development of software (e.g. to prepare model runs and to evaluate outcomes). Therefore, strict quality criterions have to be applied throughout the entire Step4 project in order to guarantee the correctness of the results. All major steps, definitions and assumptions made have to be described appropriately the keep the study transparent and traceable.

In the following subsections exemplified spatial approaches are briefly discussed for refinements of spray drift depositions and runoff entries.

2.3.4.1 Spatial approaches to estimations of spray drift deposition

Estimations of drift entry and subsequent Predicted Environmental Concentrations (PECs) may be the least complex of the three potential entry routes (drift, runoff and drainage) to examine using landscape level information, as the quantification of the spray drift deposition for certain environmental conditions can be based on an agreed empirical database (BBA, 2000, FOCUS, 2002). There are relatively fewer landscape factors affecting drift depositions, primarily the distance from crop to water and the density of crop proximate to water. Other factors such as intervening vegetation, wind speed, wind direction, humidity, etc may also be included if desired. In most cases, drift entry from standard tables (BBA, 2000) is modified based on relevant landscape factors. Subsequent PECs can be calculated based on actual or estimated water body characteristics, primarily width and volume.

The FOCUS Drift Calculator (FOCUS, 2003) implements integrated drift (using the BBA drift tables) across the width of the water body, based on inputs for crop type, water body type, water body characteristics, and specified distances between crop and water. This calculator can efficiently create water body concentrations based on those specifications and user defined application rates / frequency, while maintaining some assumptions about deposition, mixing, etc.

Many of the inputs needed to calculate the final concentrations can be derived from landscape level data as described in the landscape factors section above. Since density of cropping, distance from crop to water, and water body characteristics can vary in the landscape, methods have been developed to implement the approach used in the FOCUS drift calculator across an entire landscape, producing a distribution of landscape-level surface water PECs.

One method of incorporating landscape characteristics into drift calculations utilizes a series of ‘margins’ at various proximity distances around the water body (e.g., 10 metres, 20 metres, etc.). These distances should be determined relative to the crop type and application method. The presence or absence (and amount if desired) of target crop contained within these margins can then be used to modify the amount of drift entry estimated for a given water body. In one case, an agricultural landscape was examined for the presence of target crop (cotton) near surface water (Hendley *et al*, 2001), and the presence and percentage of target crop in each of several bands (0 – 60 metres, 60 – 120 metres, etc) was used to modify the standard the drift deposition rates (ground and aerial applications in the US) from each of these bands onto the water body (Travis *et al*, 2001). In another case, five separate margins out to 50 metres were used to describe the potential for spray drift loadings at various distances for citrus crops in Sicily, with corresponding drift estimates computed for each surface water segment (Padovani *et al*, 2004). This study divided surface water into River Assessment Units (RAU), with the goal that that each RAU had a consistent distance from crop to water and consistent width. The crop distance and water body width measurements for each RAU, along with other inputs on water body characteristics and buffer composition, were used to estimate drift loadings for each segment of the river. The drift loadings for each segment were then used as inputs to TOXSWA to calculate final concentrations at the downstream end of the water body near treated fields. It was assumed that all crops in the study area were treated, all crops were treated simultaneously, and that the wind was always blowing perpendicular from crop to water (i.e., no influence of wind direction). The use of a series of increasing sized margins around surface water allows for a straightforward and efficient method of characterizing the potential for drift entry (i.e., distance from crop to water) based on the landscape. The choice of margin distances, and the spatial definition of a water body (the unit of analysis) have an important impact on the interpretation of the results, as only a single or small number of distances and/or crop density values are used to calculate drift for each water body (or unit of analysis). This method is conservative in that the assumption is maintained that the wind is always blowing from the closest crop towards the water body (i.e., no examination of the effect of wind direction).

Another method of implementing drift calculations at the landscape level starts with a water body maximum PEC for individual water bodies, and modifies this maximum based on how much of the perimeter of the water body is exposed, and to what degree the perimeter is exposed (Holmes et al., 2002). This approach uses the same regression fit as the FOCUS drift calculator (integrates across the width of the water body) and computes the maximum drift (and resulting PEC) based on crop type, application rate, and water body characteristics (width and depth). Assumptions about wind speed and dilution are not modified.

A maximum PEC in a given direction results from drift coming from crop directly adjacent to a water body. A PEC less than the maximum will result if either (a) only a portion of the water body is potentially exposed to spray drift from that direction, or (b) if crop is not directly adjacent to the water body. This method uses two ratios called the *affected ratio* and *drift ratio* to estimate these two factors. The affected ratio estimates the portion of the water body that experiences spray drift from that direction; the drift ratio estimates the percentage of drift onto the water body in relation to the maximum.

To establish a spatial link between the water body, proximate crop, and wind direction, sampling points are placed at regular intervals on the water body perimeter. Each sample point represents a specific length of perimeter, i.e., if sample points are separated by 10-metre intervals then each point represents10 metres of water body perimeter and it is assumed that drift calculated at that point is the same over the whole 10 metres. Once points are placed, a line is drawn against the direction of the wind.

A similar approach was used by Trapp et al. (2003) to characterize the drift PEC values related to vine production in South-western Germany. Sample points placed every 10 metres along the water body were used to calculate direction and distance to vineyards. These distances were used in conjunction with the Rautmann tables (i.e., BBA drift tables, BBA 2000) to calculate deposition for each of 8 directions.

To calculate a final PEC, Holmes et al (2002) used the maximum PEC, modified by the amount of water body exposed, and the degree of exposure. The number of points at which exposure of the water body could potentially occur were counted. This number was referred to as the number of potentially exposed points (NPE). The number of lines that could actually expose the water body - because the line intersected crop somewhere along its length - were counted and this number was called the number of actually exposed points (NAE). The affected ratio is given by the ratio of the number of actually exposed points to the number of potentially exposed points (i.e. NAE / NPE).

The drift ratio with the wind in a given direction is calculated as the total calculated drift divided by the total maximum drift (i.e., the portion of the water body perimeter that does receive drift receives on average x% of the maximum). The final PEC is then calculated by multiplying the maximum PEC by the affected ratio and drift ratio.

$$\text{Final PEC} = \text{maximum PEC} * \text{affected ratio} * \text{drift ratio}$$

For a hypothetical water body with a maximum PEC of 40 ng/L, of which 60% of the perimeter is affected by spray drift, and the average drift is 50% of the maximum, the final PEC would be

$$PEC = 40 \text{ ng/L} * 0.60 * 0.50 = 12 \text{ ng/L}$$

In words, the above equation may be interpreted to read: with the wind in this direction, 60% of the exposed perimeter receives drift and that portion of the perimeter receives 50% of the maximum drift. The PEC is therefore equivalent to 30% (0.60 * 0.50 = 0.30) of the maximum value for that water body (40 ng/L), resulting in a computed PEC of 12 ng/L. Note that the affected ratio and drift ratio may both be 1.0 if crop is directly adjacent to the water body and all potentially exposed points are actually exposed for a particular wind direction, yielding a PEC corresponding to the maximum value.

Another approach to estimating drift from cropped fields to surface water uses a field-based unit of analysis and a raster GIS (Gutsche, 2001). In this approach, both land cover and surface water are in a raster format, with 5-metre grid cells. Gutsche demonstrates how easily raster analysis (which algorithms are basically implemented in up-to-date GIS) can be employed to quantify the proximity of the field to surface water. He uses the Euclidian distance between each field cell (5x5 metres) to the nearest section of a water body. Calculation of the spray drift potential of each cell is done as a function of its distance to water body grid cells using the standard drift tables (BBA, 2000).

Potential pesticide load in water bodies associated with farmed land describes the potential of absolute pesticide load per application (expressed in grams) that can come from an field to the closest water body using a dosage rate of one kilogram per hectare.

Gutsche also employs a method to calculate the potential concentration in water bodies using the same data sets. In this approach, similar to others described, the surface water is divided into units of length based on morphology and the source vector data. The calculation of the potential concentration in each water body segment is based on total load (from fields with assumed application rate), length, width, and depth of the water body.

It is important to note that in these approaches, drift loadings and subsequent concentrations are a single point in time, assuming instantaneous mixing, no upstream influences (dilution) or other temporal factors. The inclusion of temporal aspects requires the consideration of further landscape factors, e.g. a more detailed characterisation of the hydrological network (e.g. by using elevation/slope data), as well as an appropriate modeling of the fate of chemicals in surface water bodies (e.g. adsorption, degradation, dispersion, etc).

2.3.4.2 Spatial approaches to runoff estimation

To date, much effort has gone into refining surface water Predicted Environmental Concentrations (PEC) from spray drift based on landscape-level information. Knowledge of landscape-level cropping & hydrology, in combination with soils, weather and slope data, can also be applied to generate spatially refined runoff estimations.

General Concepts

The following parameters are commonly identified as affecting surface runoff: precipitation, interception, evapotranspiration, and leaching. The general relationship is:

Surface runoff = Precipitation – Interception – Evapotoranspiration –Leaching -ΔStorage

Landscape-level estimation of surface water runoff can incorporate all or some of the above components and can yield water body pesticide concentrations (PECs) or pesticide mass, or percent runoff.

- In-stream pesticide concentrations are affected by:
- The amount of pesticide introduced from applied areas
- The amount of mitigating land cover between crop and water
- The amount of non contributing (non-crop) runoff
- The amount of crop and non-crop contributions of pesticides and runoff from upstream

Data Requirements

Base data layers required for most models would include: hydrography, land cover, soils, and precipitation. Optional data for more refined analyses include may include slope derived from elevation, and a basin-scale analysis would require catchment boundary data.
Several sources of precipitation data exist. Meteorological station data is an obvious choice, however, given availability and the time and effort required to process individual station data, other sources may be preferable. FOCUS scenarios use averaged rainfall data specific to season and year to model the runoff based on crop type. Alternatively, specific ‘design storms’ may also have been created for various parts of Europe (as has been done in the US).

Predictive Models

There are several levels of complexity (and therefore accuracy) of predictive models currently being used to estimate pesticide runoff. One example of the potential incorporation of landscape-level information for runoff can be found in the Organisation for Economic Co-operation and Development (OECD) Pesticide Aquatic Risk Indicators Project (1998 – 2003) of which an outgrowth is the mechanist indicator Ratio of Exposure to Toxicity (REXTOX), which estimates spray drift and runoff. The spray drift component uses an equation based on Ganzelmeier/BBA drift values. The runoff component uses a variety of concepts and western European environmental indicators. The amount of pesticide calculated assumes rainfall three days after application. (Report of the OECD Pesticide Aquatic Risk Indicators Expert Group, April 2000).

The OECD predictor includes three additional components (correction factors) to first tier methods, describing the modification of runoff with supplementary landscape factors of slope, interception, and buffer zone. However, OECD also requires elevation data, which can be difficult to obtain and process.

In general, the runoff estimation can quantify the relative amounts of:

- Soil, land cover, and slope (to determine amount of runoff per unit area)
- Cropped areas and application rates (to determine crop runoff)
- Interception and buffer width and composition (to determine runoff mitigation)
- Non-contributing area (to determine non-crop runoff)

Table 2.3-1: Sample input parameters for pesticide runoff estimate highlighting spatial data (in yellow)

Parameter	Description	Data Sources
$L\%_{runoff}$	Runoff pesticide concentration	Resultant
Q:	Runoff	Soils
P:	Precipitation	Rainfall data
f:	Correction factor	
f_1:	Slope	Elevation data
f_2:	Interception	Land cover
f_3:	Buffer zone	Buffer Analysis/Proximity Index
$DT_{50\ soil}$:	Half-life	Generic value
Kd:	Ratio dissolved/sorbed	Resultant
K oc:	Sorption coefficient	Generic value
%OC:	Soil organic carbon content	Soils
Pc:	Stream pesticide concentration	Resultant
Ad:	Application dose	Generic value
Discharge:	Mean 'storm' stream flow	Stream gage data
ΔT:	Mean 'storm' duration	Rainfall data

Note: Parameters in yellow can be obtained from landscape-level spatial data

It is important to note that there are many methods of estimating runoff, in general and for pesticides specifically, and the above example is only meant to illustrate the concept, not to imply that only one way of runoff estimation is appropriate.

General predictor complexities and the level of effort required for data acquisition and spatial processing range in complexity from assuming a static water body and no non-contributing runoff, to more complex predictors including contributions from upstream and runoff from everywhere in the basin.

Low	• Static water body.
	• Instantaneous, in-stream discharge.
Complexity	• Instantaneous runoff from close-proximity land surface.
High	• Runoff from more distant reaches of the watershed, need time or distance.
	• Includes inputs from upstream (loadings or dilution).

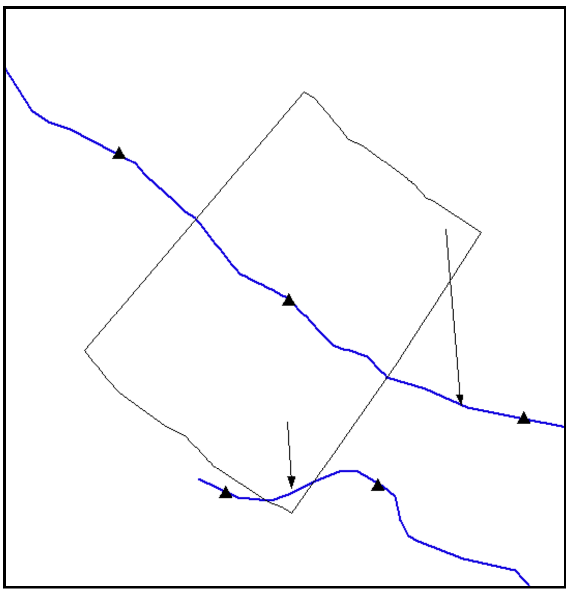
Unit of Analysis

Generally, results from estimations of drift and runoff are presented by water body, however, the spatial processing and computation is usually performed at a field-level and as such, the spatial relationship between land surface and water body is critical to the unit of analysis, which includes both the water body and the surrounding area to be included in the analysis.

Water body: A water body can be defined several ways, most commonly confluence to confluence or segments of equal length. The standard water body is a 1-meter wide, 0.30-meter deep ditch, but can be modified according to spatial location (for instance may not be appropriate for southern EU areas) or local information. Because water runoff is usually included in the estimation of pesticide runoff, the direction of surface water runoff needs to be taken into consideration. The spatial extent of analysis, the margin, is therefore limited to a certain extent by the minimum length of a water body.

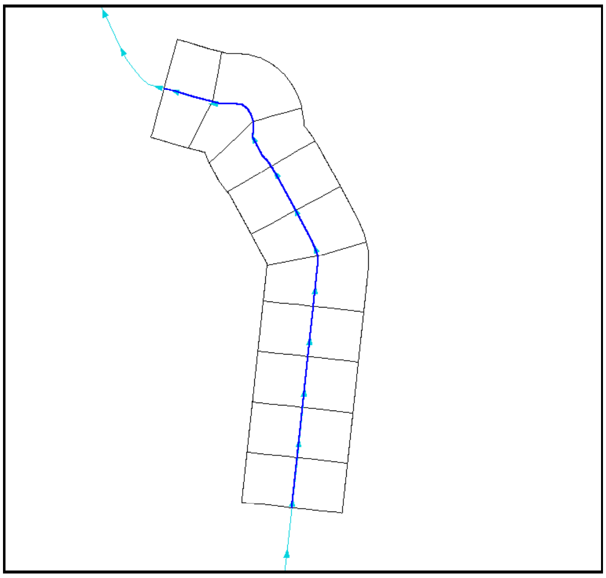
Margin: Margin distances should be scaled to water body length and topography such that runoff generated at the outer edge of the margin is likely to flow into the margin area, i.e., the runoff flow direction is reasonably orthogonal to the channel (Figure 2.3-4). For example, if the average stream segment is 1 kilometer, the maximum appropriate margin distance may be 200 meters, depending on local topography.

Figure 2.3-4. Margin extent should be scaled to water body length



Segment: The water body (confluence to confluence) can be further subdivided into *X*-meter segments (e.g., 100 meter), to provide more spatially refined exposure estimates (Figure 2.3-5). This spatial refinement can be applied to all landscape level metrics generated, such as drift PECs, margin composition, and buffer widths. Therefore, this portion should be considered separately from the runoff component, as it can be applied on a more widespread scale if desired.

Figure 2.3-5. Example 100-meter segment processing with 100-meter margin

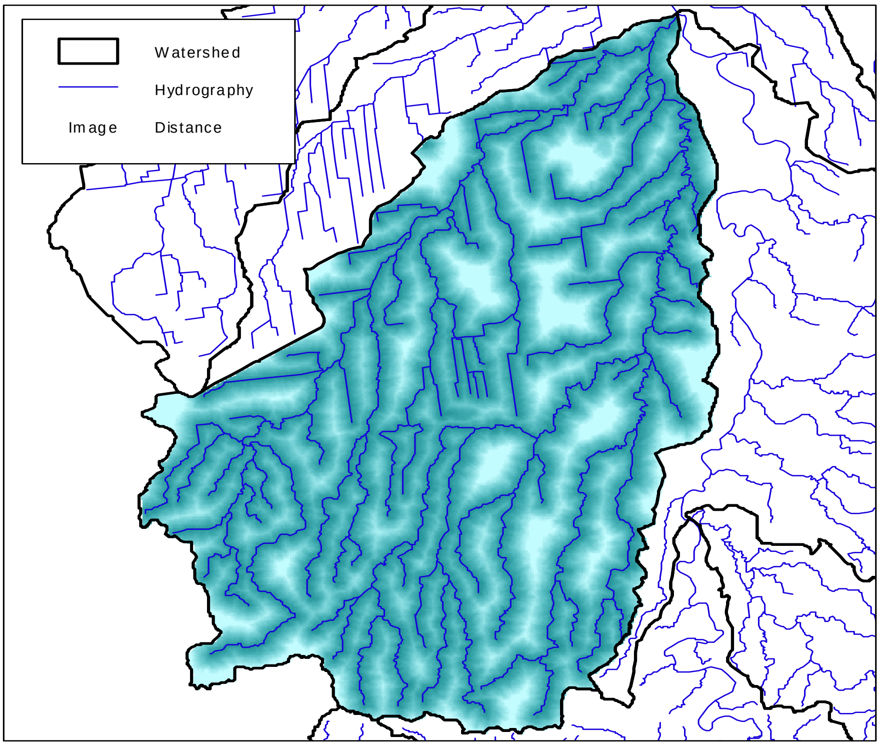


Basin: There is also the potential to estimate runoff at the basin scale, however there are several caveats to this level of processing. The temporal aspect of runoff becomes a component that must be addressed in terms of estimating runoff from the furthest reaches of watershed, as the concept of instantaneous runoff is no longer applicable at the basin scale. Two suggested solutions to incorporate the temporal aspect are:

a) The use of multiple margins within which the distance between the margin and water body will serve as proxy for the time for runoff to reach the channel, or

b) A continuous distance coefficient using the actual distance of crop to water throughout the watershed (Figure 2.3-6)

Figure 2.3-6. Sample showing continuous distance coefficient using actual distance from entire catchment (50-m pixels) to nearest down slope channel



2.3.5 Relating landscape-level results to a larger area

Refinement of the risk assessment process from Step 3 to Step 4 can include the use of spatially referenced landscape level information. While this information allows for a better understanding of the agricultural landscape, it is important to also understand how that particular agricultural landscape (selected for specific crop, environmental or other factors) relates to the broader EU registration process and demonstrated safe use. While the scenarios selected by the FOCUS Surface Water workgroup are meant to be representative of very large regions within the EU, the scenarios themselves, limited by their number, must use some broad characterizations of parameters. If these parameters are refined based on more landscape level spatial information, the ability to place the refined spatial information (and derived results) into a broader context is critical.

In a simple case where drift is the primary concern, and a study area was selected and analysed spatially to refine particular model inputs, it must be understood where this study area fits within the characteristics affecting drift on a broader scale (agricultural density, hydrologic density, and crop proximity to surface water). Drift was selected for this discussion because potential exposure is driven primarily by cropping density and proximity, but similar concepts apply to runoff and drainage entry using additional landscape factors (soils, slope, precipitation). See Volume 1 Appendix A1 for a full illustration of this process.

2.3.5.1 Grid approach

In many cases, given the large area to be examined in the context setting process (entire EU, north or south EU, FOCUS scenario, ecoregion, member state, etc.) a gridding process is commonly used to provide a spatial "unit of analysis" at a specific level. The size of the grid can vary from as small as 1km, up to 10km, 25km or 50km depending on the area to be covered and the input data sets available. For each grid cell, the relevant data are collected and used to determine relative (or absolute) potential exposure within the broad overall area (e.g., the northern EU). Potential exposure for the grid cells in the area selected and analysed in detail are also identified. These grid cells (which have been studied in detail using more refined data sets) can then be placed on the distribution of entire grid cells for the larger area (such as the northern EU). The location of the grid cells for the study area on the overall distribution give confidence that the study area (and the associated landscape characteristics and derived modelling parameters) represent the desired goal ("normal" case, conservative case, 90th percentile, etc.).

As an example, one method of the grid approach uses 10km grid cells across the EU to quantify three metrics related to potential aquatic exposure from arable agriculture:

1. Percentage of grid cell composed of arable crop,
2. Percentage of grid cell composed of surface water, and
3. Percentage of grid cell composed of arable crop within 1500m of surface water,

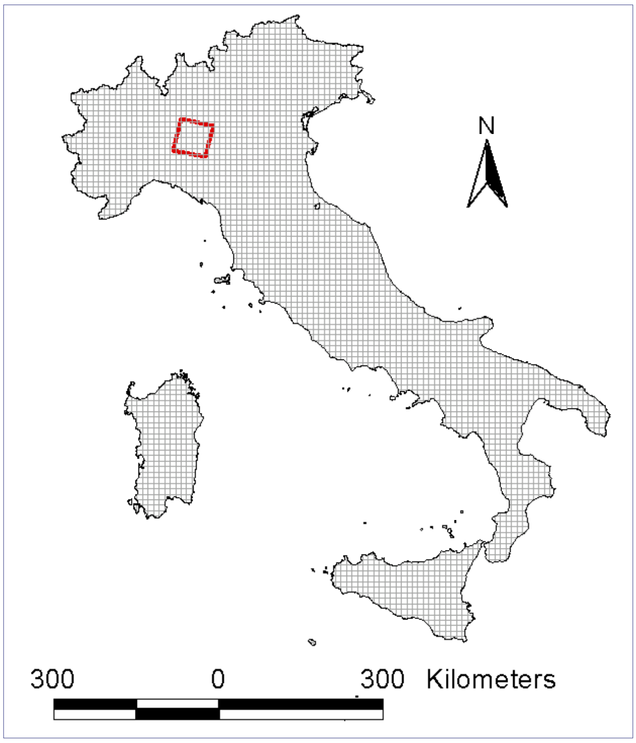


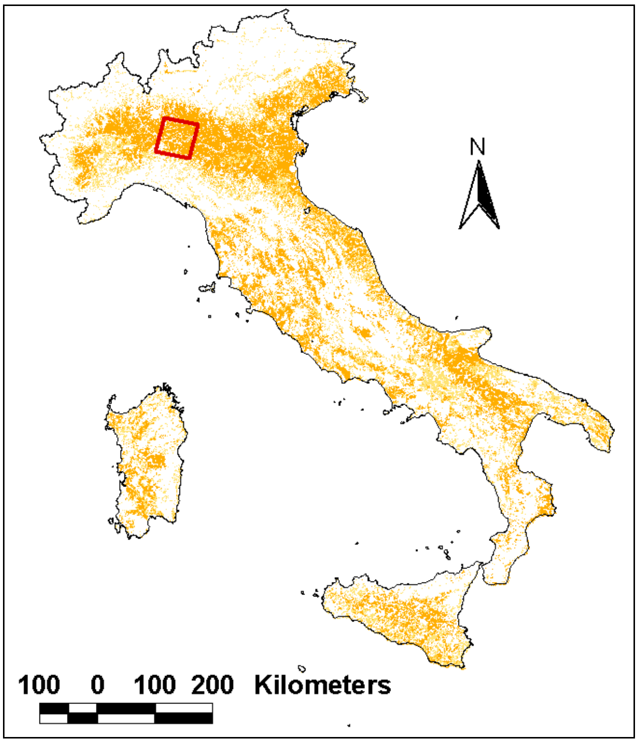
Figure 2.3-7: Grid of 10km cells covering Italy

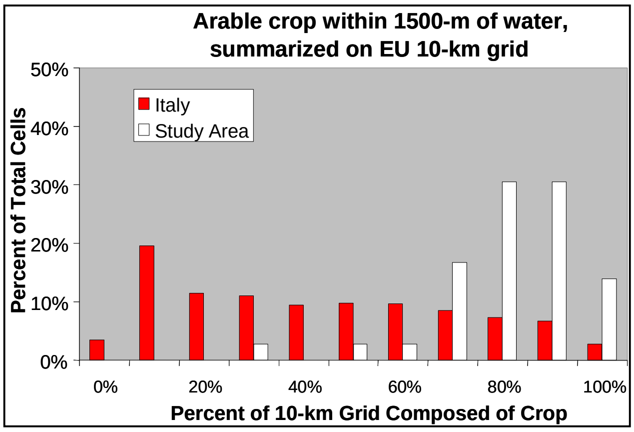
The 10K grid cell provides a unit of analysis for EU and MS review which is representative of the scale of agricultural landscape.

For this example in Italy, the number of 10-km grid cells for Italy is 3,238 and the study area (roughly 60km x 60km) contained 36 cells.

The CORINE land cover was used to determine the location of arable crop, and potential surface water locations were derived from GTOPO30 1-km elevation data. The percentage of the total area within each grid cell comprising each of these was recorded. In addition, the amount of arable crop located within 1500m of surface water was also calculated. This single metric provided a reasonable estimate of potential surface water exposure to arable crop at the coarse EU or Member State level, used to identify areas for further, more detailed, examination. Distributions of this "potentially exposing crop" are shown in the following two figures, both spatially and graphically. Note that the histogram contains data series for both all of Italy and for the specific study area.

Figure 2.3-8: Arable crop in Italy, and distribution of crop near water density for Italy and study site.





It can be seen that the study area contains a higher proportion of arable crop within 1500m of surface water than the larger area examined (in this case, Italy). Therefore, more detailed landscape-level analyses performed in this study area, and specifically for several of the 10km cells, represent an upper end case for potential exposure to surface water from agricultural influences (arable crop).

2.3.5.2 Administrative unit approach

An alternative method to the gridding approach is to use cropping statistics and some level of administrative unit. In this approach, several administrative areas (NUTS 4 or 5) are studied in detail to generate landscape level modelling inputs. General factors affecting potential exposure are then summarized for each administrative unit (cropping density, hydrological density, etc). The entire set of administrative units, including those studied in detail, are then ranked according to potential exposure. The location of the administrative units for the study area on the overall distribution give confidence that those studied in detail (and the associated landscape characteristics and derived modelling parameters) represent the desired goal ("normal" case, conservative case, 90th percentile, etc.). An advantage of this method is that cropping information may be more available in statistical form at the administrative unit level (rather than in spatial form), it allows for a quantitative statement about the percentage of the entire crop production analyzed (e.g., 15% of all maize production in the member state was analyzed in detail to produce the landscape level modelling inputs), and that crop sub-types (such as maize instead of 'arable') can be examined. Drawbacks to this approach include difficulty in implementation across member state boundaries where crop statistics can vary, and using administrative boundaries to define the agricultural unit of analysis (which vary in size and are generally not delineated with agricultural and even environmental factors in mind).

Keep in mind that methods of applying specific crop sub-types (such as maize instead of 'arable') exist to place the crop type information from the statistics "on top of" spatial land cover information. See subsequent section for more information.

2.3.5.3 Catchment area approach

Another method uses catchment areas (or basins) as the unit of analysis. As with other methods, cropping density, hydrological density, or other relevant factors are examined at the catchment level. The location of the catchment(s) contained in the study area on the overall distribution give confidence that the study area (and the associated landscape characteristics and derived modelling parameters) represent the desired goal ("normal" case, conservative case, 90th percentile, etc.). This method has the advantage of using a hydrologically relevant unit for surface water, knowing that concerns related to surface water are more likely to be homogenous within the catchment than between catchments. Drawbacks to this approach include difficulty in obtaining catchment boundaries of sufficient scale and extent, especially between member states. Also, cropping density and production must be generated from spatial land cover data, as cropping statistics are not reported by catchment area (see next section).

2.3.5.4 Combining crop statistics and spatial land cover information

In many cases, land cover information does not identify specific crop types that may be of interest. For example, the Corine Land Cover data set includes several classes for arable crops, but does not sub-classify individual crops or crop groups. Therefore, an examination focusing on maize but using a spatial land cover layer identifying only arable crop would have to make an assumption that maize production is evenly distributed over all arable crop areas.

To partially address this, crop statistics (usually at some administrative unit level) can be used to calculate the ratio of maize production to arable crop production. In other words, density of maize production (of all arable crop) for each administrative unit can be calculated. This maize density for each administrative unit can then be used to identify the geographic areas where the arable class from the spatial land cover data represents greater amounts of maize production. This information can also be used to "allocate" the maize production (in hectares) to other spatial units, such as catchments or grid cells using the spatial land cover data. This results in a quantification of maize production (in hectares) for spatial units other than those originally reported in the crop statistics. Region level crop statistics will be used to illustrate this example, even though better resolution crop statistics (canton level) exist for maize production.

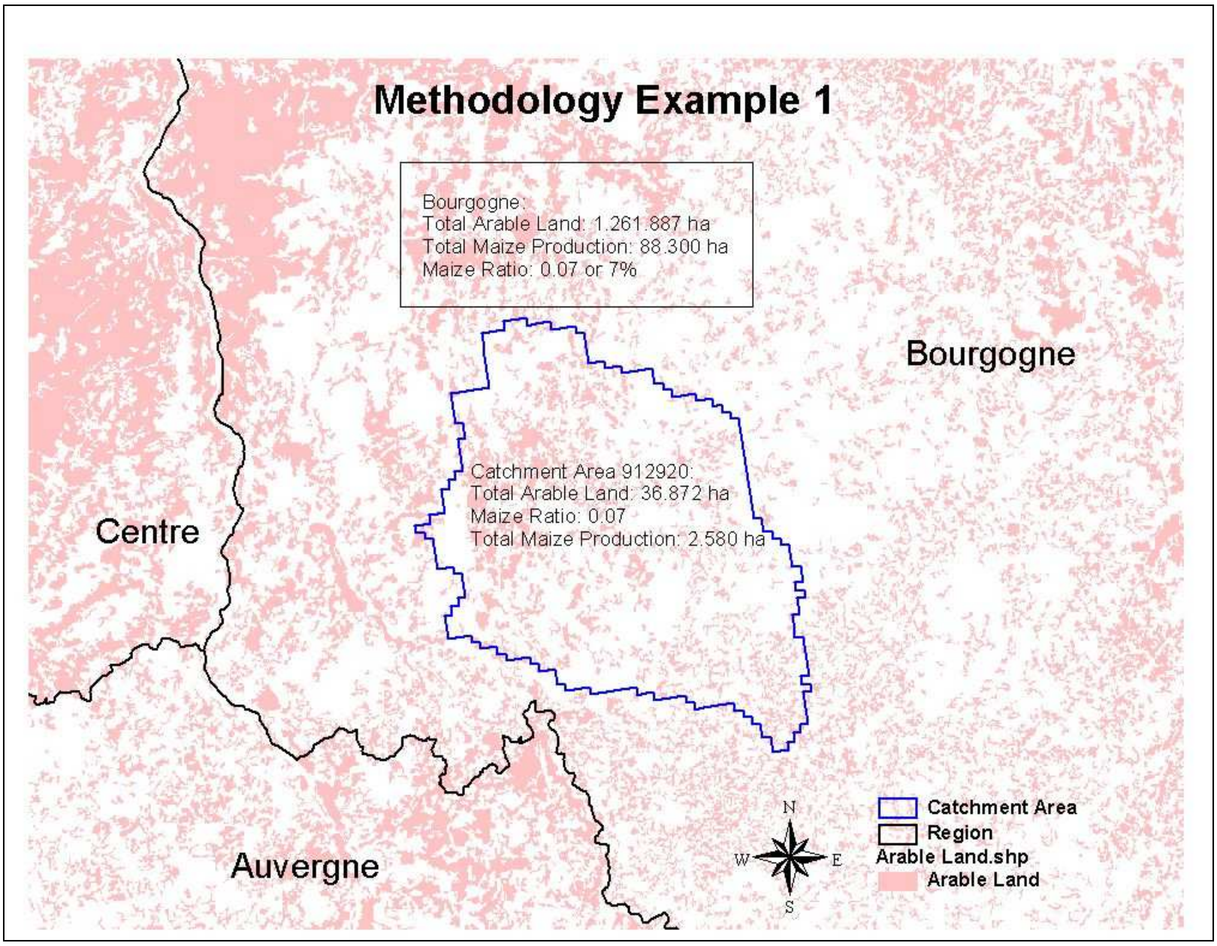
The general methodology can be divided in to the following steps using the maize example, the Corine land cover, crop statistics for France, and Hydro1K catchments:

1. Assign the statistical production numbers to the administrative units (NUTS 2, Region) in a GIS;

2. Determine the amount of arable crop land within each administrative unit (Region) using the Corine Land Cover data set;
3. Calculate the ratio between total crop area and maize area for each Region (the 'maize ratio');
4. Spatially intersect the administrative units and the catchment areas;
5. Determine the amount of arable land within each catchment area, and apply the maize ratio to estimate the amount of maize in the catchment area. If multiple Regions are found within a single catchment area, process each separately and combine the maize area for each Region into a single result for the catchment area.

The figure example illustrates a simple example in which a catchment area (#912920) is wholly contained within a single Region.

Figure 2.3-9: Example of combining crop statistics and spatial land cover information

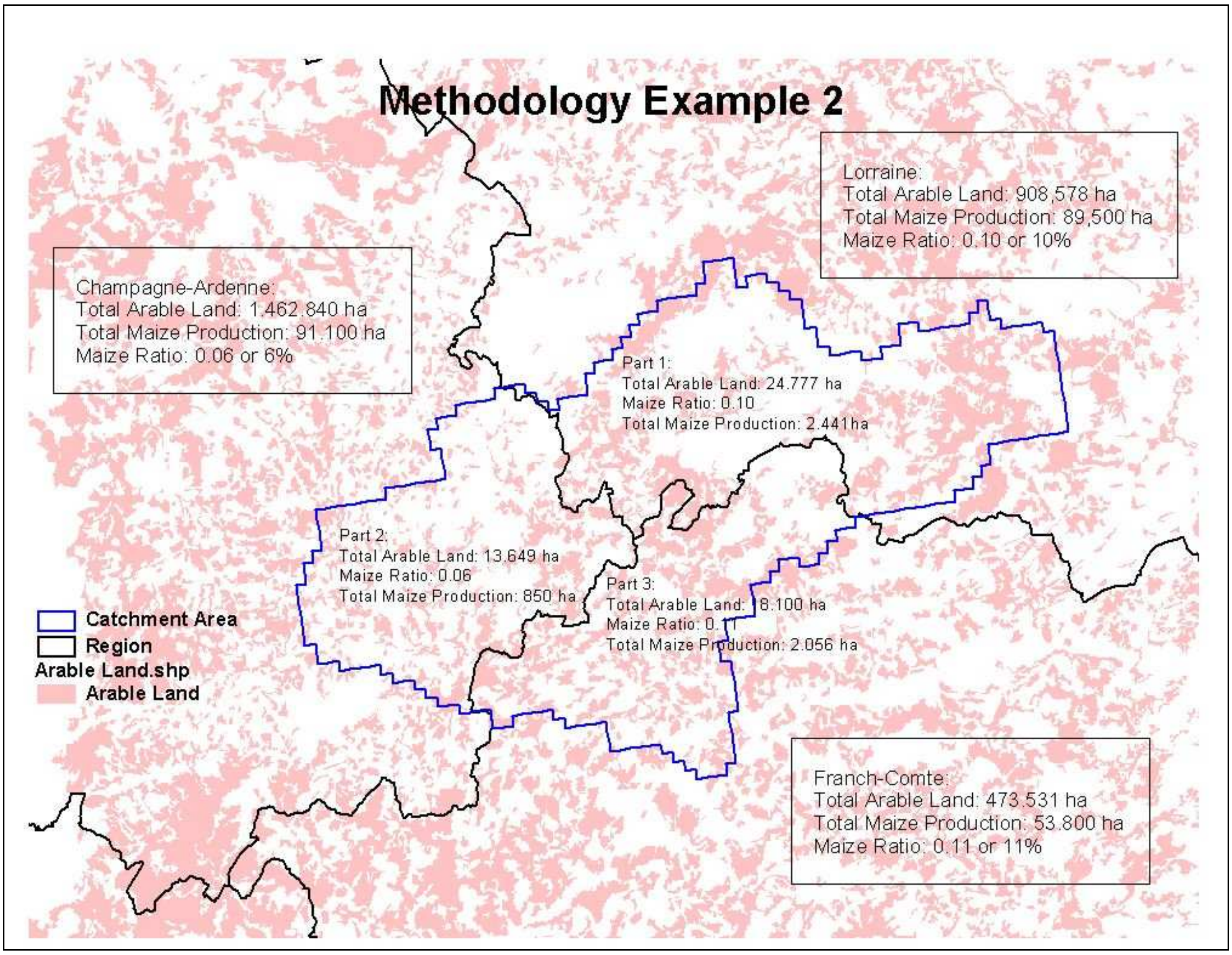


In this example, the Bourgogne Region produced a total of 88,300 hectares of maize in 1997. This Region also contains 1,261,887 ha of arable cropland as defined by the Corine Land Cover data set. The maize ratio is computed as the hectares of maize divided by the total arable land, resulting in 0.07, or roughly 7% of the arable land is maize for this Region.

The catchment area (#912920) contains a total of 36,872 ha of arable land (as defined by the Corine Land Cover data set). Because this catchment area is completely contained within Bourgogne, it is estimated that 7% of this arable land is cropped with maize, resulting in 2,580 ha of the total maize produced in the Region is attributed to this catchment area.

The next figure illustrates a more complex example, in which the catchment area contains parts of three different Regions.

Figure 2.3-10: Second example of combining crop statistics and spatial land cover information



In this example, the catchment is divided among three different Regions, each having a different maize ratio. The catchment is partially located in Lorraine (0.10 maize ratio), Champagne-Ardenne (0.06 maize ratio) and Franch-Comté (0.11 maize ratio). Maize production is estimated for each of the three parts individually, and a total is computed from these results. For example, Part 1 is located in Lorraine, contains 24,777 ha of arable land, has a maize ratio of roughly 0.10, and therefore contains a total of 2,441 ha of maize. Likewise Part 2 (Champagne-Ardenne) contains 850 ha and Part 3 (Franch-Comté) contains 2,056 ha. Therefore, the entire watershed contains 2,441 + 850 + 2,056 = 5,347 ha of maize.

The approach of using crop statistics and general land cover data to refine the spatial location of the crop production can be utilized at various scales. Whenever possible, the highest resolution of crop statistics and land cover should be used.

2.3.6 Approaches for generating landscape factors used as supportive information for higher tier exposure assessment

Spatial tools available for landscape analysis can yield valuable information about the interaction between agriculture and surface water. In some cases, this information is not a direct input to the existing modelling process. While this information is quantitative in nature (being derived from spatial and statistical information), it currently only provides a qualitative factor in the risk assessment process. This kind of information can include the total amount of crop and non-crop area, the number of water bodies and total amount of surface water area located within x metres of crop, the total amount of crop area located more than x metres from any surface water, etc. These types of information provide a valuable insight into the understanding of a particular agricultural landscape, whether it relates to specific FOCUS scenario, a relatively large area (e.g., 360,000 hectares) studied in detail, or more targeted areas based on specific crop or usage criteria.

Discussion in this chapter assumes that a geographic area has been selected with specific criteria, and studied in detail using spatial approaches. For example, if the issue were drift entry related to cereal production in northern climates, a suitable location would be selected and examined using appropriate data. Likewise, if runoff issues related to citrus production in southern climates were of concern, a representative area would be selected. In each case, the location and total area selected for examination will be dependent on specific criteria, and all cases cannot be covered here. Annex A4 of Volume 1 illustrates one approach used for this process.

However the specific area is chosen, several factors can aid in the interpretation of the Step 4 approach taken. These factors are discussed in the following sections.

2.3.6.1 Overall cropping intensity & distribution

Information comprising total area studied and the total amount of crop within that area give an indication whether this area is representative of an intensive agricultural area. For example, if 300,000 hectares of land area were studied, encompassing over 200,000 hectares of crop, it could be argued that the study area represents a relatively intense agricultural area. Ideally, if data permit, crop statistics will be incorporated to further describe the importance of the study area in relation to cropping. For example, if the above 200,000 hectares of crop represent 25% of the total production in a much larger area/region, further confidence is added that the selected study area exemplifies an intensive agricultural area, and that the landscape-level modelling results derived from these data can be considered better substantiated.

The general distribution of crop over the landscape may also provide insight into potential exposure. For a given amount of crop, are small, intensive cropped areas distributed in clusters across the landscape, or is there a broader distribution of less intensive cropped areas? If the former (small intensive clusters), it would be expected that potential exposure to surface water would have a greater range (from relatively non-exposed up to significant exposure); whereas a more even less intense cropping pattern might indicate exposure with fewer extremes (both low and high)

2.3.6.2 Overall water body type and distribution

The frequency and relative contribution to the hydrologic network of different water body types is an important factor to understanding the potential exposure to surface water. Water body types are generally separated into classes such as streams, rivers (essentially wide streams), canals and ditches (man made for transport of water either to or from a field) and static water bodies (ponds, reservoirs, lakes). In some cases, the permanence of the water body is also tracked. Understanding the relative contribution of each of these water body classes (in both frequency and length/area) helps to place the modelling results into context within the hydrologic context. Water bodies may also be classified according to chemical and biological status of water quality if these data are available and have a geographic link to the hydrologic network (i.e., a latitude and longitude value for surface water monitoring sites, strem reach identifier, or even a stream name if this also appears in the spatial data).

Using our example of a 300,000 hectare study area, it might contain a total of 5,000 water bodies, with specific numbers associated with each class. In addition, the total length of all flowing water (perhaps 6,000 km) and for each water body class can also be reviewed. Similarly, numbers and area for ponds and lakes can be reported (including a distribution for different pond size classes). Two qualitative factors can be assessed from this information. Firstly, that a significant number of water bodies (in either count or length/area) have been assessed in the landscape to produce the modelling inputs; and secondly, that the landscape analysed represents not only an intensive agricultural area, but also one that relates to the specific surface water body class of concern.

2.3.6.3 Water bodies within a specific distance of crop

Once the general landscape of the entire study area is understood, more specific questions can be answered about potential exposure of surface water to crop at the landscape level. One of the most basic is reporting the number of water bodies that have any portion within a specific distance of crop. The distance used for this will depend on the method of transport (drift, runoff, drainage) and possibly crop type (for example, larger distances for crops sprayed with air blast sprayer rather than boom sprayers). In addition, since a stream or a pond is not an atomic unit (not either wholly effected or unaffected), the amount of surface area within a given distance to crop can be useful.

For example, using the set of 5,000 water bodies in the 300,000 hectares studied, it could be reported that overall, 75% of the water bodies were within 100m of crop. This could be further refined to show that ditches are more likely to be within 100m of crop (85%) than streams (60%) and ponds (70%). Furthermore, while 75% of all water bodies are within 100m of crop, this accounts for only 40% of the total surface water area (with similar differences between water body classes as seen earlier). Finally, the amount of potentially exposed surface area for each individual water body can also be determined and the resulting distribution reported. Information about the landscape at this level provide a background for the assessment of the modelling results. For instance, the modelling results may indicate a concentration when a water body is (maximally) exposed, but the landscape information may indicate how often/likely a water body may be exposed.

2.3.6.4 Amount of crop located outside a specific distance of surface water

If it can be determined that only crop within a certain distance has the potential to negatively effect surface water (this may only apply to drift and runoff), it is important to understand how much of the crop in the study area is within this distance. While this cannot be used directly as a model input, it does give a general understanding of how much of the crop that is there, may impact surface water.

For example, out of the total 300,000 hectares in the study area, 200,000 hectares of crop were identified. From this total area, it could be determined that 25,000 hectares (12% of all crop) is located within 100m of surface water, or stated inversely, 88% of all crop in the study area (175,000 hectares) are located more than 100m from any surface water. In other agricultural landscapes, the amount of crop proximate to water might be quite different, perhaps 65% of all crop is within 100m of water. Knowing which of these agricultural landscapes are being studied is crucial to evaluating the Step 4 modelling inputs and results.

2.3.6.5 Crop variation

While intensity of cropping near surface water is a direct indicator of potential exposure, knowledge about the variation within the crop class can provide important information that can be used to modify the potential exposure. If an area is heavily cropped with only one or two specific crops (monoculture), the potential exposure to surface water from applications made to those crops is most likely greater than to those same water bodies if the cropped areas had a much greater diversity of crops, due to the greater likelihood of variations in applications.

For example, if the 200,000 hectares of [arable] crop in our example are primarily composed of only maize, there is a greater chance that applications of pesticides will co-occur in time due to similar planting/maturity dates, possibility of insect infestations, etc. On the other hand, if that 200,00 hectares is composed of a variety of arable crops (maize, sunflower, sugar beet, winter cereals, OSR, etc), there is less likelihood that surface water in the area will impacted as much as with the monoculture example, due to variations in application timing and location.

Crop variation can be assessed in an area using several methods. The easiest approach is to use detailed crop statistics to review the diversity of crops grown in the area studied (and their relative production). While this method provides a quantitative approach, it relies on statistics being reported at a sufficiently detailed resolution (i.e., NUTS 4 or 5). It also may not provide sufficient information regarding the spatial distribution of the crops. In other words, while a given area may have a diversity of crops, the crops still may be grown in concentrated groups within the overall area, resulting in potentially higher exposure to the water bodies in those areas.

Detailed spatial land cover data can provide information on the actual distribution of crops (at a single point in time), and their relationship to each other. For example, it could be determined if the variety of crops in the area tend to be mixed together, with fields having alternating crops, so that the potential exposure to any given water body in the area will likely come from a variety of crops, possibly with a variety of application times and needs (products, methods of application, etc). Keep in mind that the spatial land cover data must have crop classes that differentiate between similar crop types (e.g., different types of arable crop). In many cases, this can be difficult to obtain, and only pertains to a single season (although historical crop rotation practices can be reviewed at a local level).

Knowledge of the spatial variation in cropping can provide supporting, qualitative information to the standard risk assessment process.

2.3.6.6 Field size variation

Similar to variations in crop types, variations in field size can be examined to gain a better understanding of the agricultural landscape. The size of a field can be an indicator of the cropping homogeneity of the landscape and the likelihood that a larger area will be treated at a given time (i.e., a 200 hectare field is more likely to be treated at a single time than four 50 hectare fields). Since fields are usually bounded by physical features (roads, hedges, trees), the size of a field is less likely to vary from season to season than the actual crop grown in the field. Field sizes may also be more accessible from spatial information provided by national mapping agencies, and also due to the increased use of spatial information systems in crop subsidy monitoring programs.

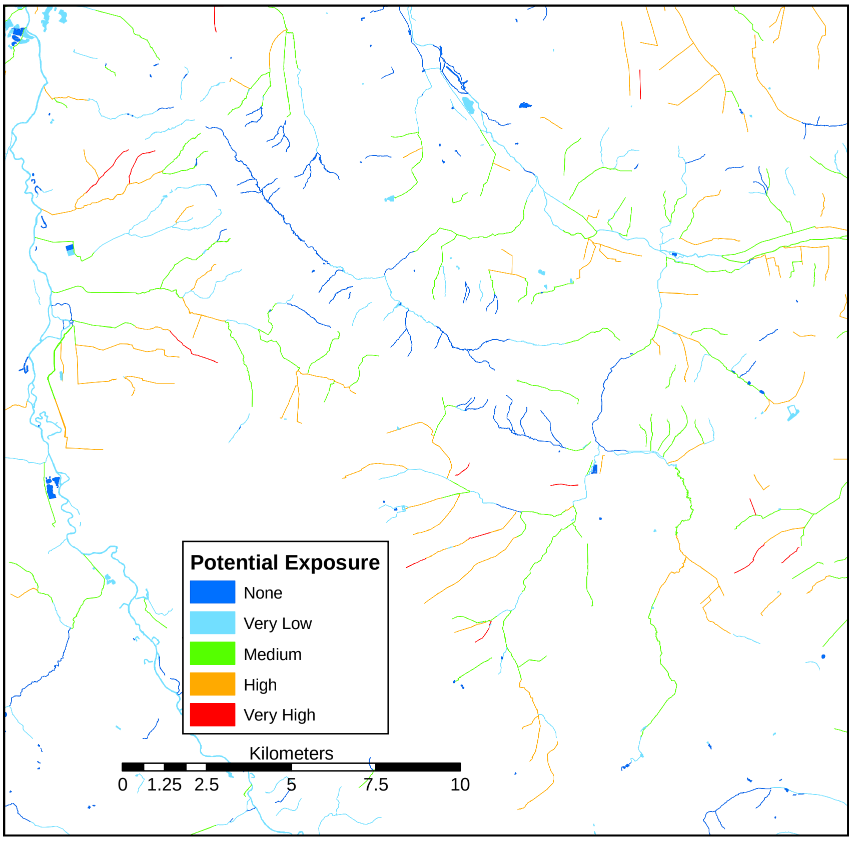
2.3.7 Evaluating the spatial distribution of results

Landscape-level analyses of surface water in an agricultural area generally produce results for each ‘unit of analysis’ (see Section 2.3.1). The unit of analysis can be a water body, an agricultural field, a catchment area, an administrative unit, etc. The “landscape” provides variation for some of the inputs used to define the potential exposure, and therefore produces a range of potential exposure values. This range of values is typically expressed as a percentile (for a single value or threshold), or as a histogram (to represent the distribution of a set of values).

The spatial influence of the landscape can also be examined in terms of the resulting distribution of potential exposure across the landscape using maps and other cartographic methods. The spatial distribution may be important to understand the potential exposure of a landscape (an agricultural area) rather than a set of water bodies or agricultural fields and discrete units (as percentiles represent them). The relative spatial relationship between greater and less exposed water bodies can be important. For example, two study areas with the same relative proportion of water bodies exceeding some value, or with the same 90th percentile value, may have two different exposure / risk potentials based on the location of those greater exposed water bodies. In one case, the 10% of water bodies above the 90th percentile might be grouped in one very intensive area, while in the second case, the same top 10% of water bodies may be scattered across the landscape, interspersed with other less or non-exposed water bodies. Because the ecological health of surface water can be influenced by connecting and/or neighboring water bodies (e.g., recolonization), the location of the waterbodies with potentially greater exposure relative to other less or non-exposed water bodies can be an indicator of the relative risk of surface water in agricultural areas similar to that examined using landscape information.

The figure below illustrates an example of the spatial distribution of potential exposure across a portion of an agricultural landscape. It can be seen that water bodies of greater potential exposure (colored in orange and red), are distributed among other lesser, or non-exposed water bodies. This spatial variation in potential exposure can aid in the evaluation of the agricultural landscape as it relates to surface water.

Figure 2.3-11: Example of spatial distribution of exposure to surface water



2.3.8 Use of remotely sensed data in landscape characterization

In conducting a landscape-based exposure assessment, it is often essential to obtain detailed information about the composition and distribution of the landscape in order to characterize it. Historically, generalizations were used because it was too difficult, time consuming and costly to acquire accurate and detailed information about the landscape. In recent years, however, advances in computer and remote sensing technology and the increased availability of remotely sensed data sources has made it possible to generate accurate and detailed landscape characterizations in a cost effective manner. While a very powerful tool in the exposure assessment toolbox, there are a number of factors that need to be considered when using this technology in order to generate the results necessary for a scientifically valid study.

2.3.8.1 General characteristics of remotely sensed data

The primary use of remotely sensed data in exposure assessment is to generate a land cover dataset of the area being analyzed. While there are currently a variety of remotely sensed data sources with their own strengths and weaknesses, in general there are certain properties of remotely sensed data that should be acknowledged and accounted for.

Reflectance

Remotely sensed data is based on the reflectance of an energy source from the earth's surface. This energy source can be visible light, infrared radiation, thermal radiation or even radar. Each type of energy source provides different information about the surfaces from which it is reflected. Because reflected information is captured by a sensor, only those things that can be seen by the sensor can be identified. For example, this means that if a water body is obscured by brush or trees, the water body cannot be identified using remotely sensed data alone.

It is important to note that for many landscape analyses, the real power of remotely sensed data is not necessarily in the visual interpretation of the data but in the information contained within the reflectance information itself. By using differences in reflectance in very detailed portions of the spectrum (red, green, blue, near infra-red, etc.), the imagery can be classified using semi-automated processes that permit accurate and cost effective land cover classifications to be generated for large areas. For example, by using the fact that vegetation is green and soil is not, we can separate soil from vegetation. By using information in the infrared portion of the spectrum, we can identify different types of vegetation and separate crop from forest, etc.

Resolution

All remotely sensed data has a resolution that is based on the sensor technology used to capture the reflected energy and the physical properties of the reflected energy. This resolution is often stated in a "pixel" size, with a pixel being defined as the smallest unit of information captured by the sensor. For example, a satellite based sensor may have a ground resolution (pixel size) of 30m x 30m on the ground. An aerial based sensor may have a ground resolution (pixel size) of 1m x 1m on the ground.

Geo-Rectification

All imagery is acquired as a snapshot that has no pre-defined link to the real world. While it obviously represents a specific location on the earth's surface, the image must be geo-rectified in order to permit integration with other types of digital data such as hydrology or administrative unit boundaries. The geo-rectification process involves identifying specific features or locations in the imagery as well as on a map, or other reference source, that provides the map coordinates for the features or locations. Once a sufficient number of reference points are identified, the imagery is mathematically resampled and placed into the map projection of the reference data. This process is also commonly called geo-referencing.

Geo-rectification is an absolutely critical step in the processing and one that can have a significant impact on the quality of the resulting land cover classification. If the points used for the rectification are selected inaccurately, the data will not overlay with other datasets correctly. Also, if the rectification is done at an inappropriate time in the processing of the imagery, it will negatively impact the accuracy of the land cover classification.

Accuracy assessment

The quality of the resulting analysis is dependent on the accuracy of the land cover data used to define the type and location of land cover types in the study area. In order to determine the level of confidence in the accuracy of the classification, an accuracy assessment must be undertaken. To conduct an accuracy assessment, ground-truth information should be acquired as close to the date of image acquisition as possible and the ground-truth information should provide information regarding the actual land cover that exists in specific locations within the imagery.

The timing of ground truth data acquisition is important. For example, if the ground truth data is acquired several months after the imagery and the crops in the ground have changed since the image was acquired, then it would be virtually impossible to verify the accuracy of a crop type classification.

Complete coverage of the imagery is not necessary, but sufficient sampling of heterogeneous areas to generate a statistically significant sample size should be performed. By then comparing the actual ground-truth against the semi-automated classification results, the level of confidence in the derived land cover dataset can be determined.

Point in time

It is important to remember that the acquisition of the image represents a single point in time. When using the land cover data generated from the image (or any land cover data for that matter), issues of crop maturity, crop rotation, double cropping in a year, and changes in crop types should be considered. While a crop field tends to remain a crop field year after year, the crop grown in the field is likely to change. Therefore, the temporal issues of land cover data based on remotely sensed imagery are related to the crops of interest, with permanent or broad crop categories (e.g., arable crop) being less impacted by a point in time classification, and specific annual crops (e.g., vegetables, maize) being more sensitive to the temporal issues.

2.3.8.2 Considerations for acquiring remotely sensed data

Regardless of the specific type of data acquired, the following are a number of general factors that should be considered when obtaining remotely sensed data for exposure assessment analyses:

Time of Year

If a specific crop is to be identified within the imagery, then it is necessary to acquire imagery at a time of year that will maximize the discrimination of that crop within the imagery. It does no good to acquire data when the crop is not in the ground, or when it cannot be differentiated from other crops or land cover types.

Resolution

It is important to select the resolution of the data that is appropriate for the analysis being conducted. For example, if a study is undertaken to identify natural landscape features that exist between cropped fields and water bodies and the typical distance between a field and a water body is approximately 10m, it would be inappropriate to use imagery that has a 30m pixel size as the data would simply be unable to differentiate any landscape parameters at a smaller than 30m scale. Said another way, the reflectance of all the different features of the landscape within the 30m x 30m pixel would be averaged into a single reflectance value that may include water, crop and trees. This would not be useful to the analysis. On the other hand, if analyzing a large area of millions of hectares and only general landscape categories are required such as cropland, forest, water, etc., then a coarser 30m pixel size would be more appropriate than using fine resolution aerial photography with a 1m pixel size.

Type of sensor

The type of imagery acquired can have a large impact on the outcome of the resulting land cover classification. If a purely visual interpretation of the landscape is required, then simple black and white or true-color photography may be appropriate. If a large area of analysis makes visual interpretation impractical, then multispectral data may be preferable. If separation of crop types or other more difficult analyses are to be performed, then multispectral data with infrared reflectance information would likely be required.

Image footprint

There are generally two types of remotely sensed imagery, satellite based and aircraft based. An important characteristic of these platforms is the area encompassed in a single image footprint. Satellite images generally have significantly larger areas of coverage in a single image footprint than do images acquired from an airborne sensor. The tradeoff is resolution, with footprint size being inversely proportional to the resolution of the data. The process of merging multiple small images into a single larger image is called "mosaicing" and can be a very costly process that impacts the overall accuracy of the resulting land cover classification. Air photos cover less area on the ground but have higher ground resolution. Satellite images have coarser ground resolution but much larger footprints. If a small study area is to be analyzed, then imagery from an airborne sensor may be more effective. If a large study area is required, then satellite based imagery may be more effective.

Availability of ancillary data

Often features can be identified in a land-cover classification and accurately located in relation to other features within the image because the entire area is being geo-rectified as a unit. This is an advantage as it permits the relative positions of all features to be accurately represented. However, you are more limited in the descriptive information that can be derived from features in remotely sensed imagery. In this case, the availability of ancillary data such as digital hydrology information may be factored into the analysis. If the types of water bodies such as streams, rivers, ponds, man-made lakes, etc. need to be known, it may be necessary to spatially link ancillary hydrologic data from another source to the water bodies found in the imagery in order to provide a complete picture for analysis.

2.3.9 General research recommendations

In order to address the growing use of spatial technologies in landscape analysis for risk assessment, several issues should be identified and addressed in future research. Because of the relative newness of these spatial approaches (as compared to other methods in risk assessment), confidence in the scientific, consistent, and ethical application of these technologies may be a concern on the behalf of regulatory agencies. In addition, since multiple approaches in the application of spatial technologies to a given problem may yield similar results, interpretation of results requires a moderate level of understanding in spatial processing to assess the relevance and validity of the methods used. This in turn may introduce hesitation on the part of the submitter to invest in the development and application of these spatial techniques if there is a possibility they will not be fully accepted.

It is therefore recommended that further research be applied to the development of a standard set of landscape factors and methods, as an available "toolbox" of accepted spatial approaches for exposure assessment. In addition, a structured set of guidelines and definitions for spatial data and the related application to risk assessment should be developed.

While there are several initiatives underway in the EU to generate and distribute spatial information in a consistent and transparent manner, most of these do not address pesticides and surface water as the primary focus. Those initiatives that focus on water quality in surface water (such as the Water Framework Directive), are commonly applied to larger river basins with multiple input pollutants than to the small streams and ponds in agricultural areas. Other initiatives that address data gathering and dissemination on the EU level (e.g., INSPIRE, GINIE, etc) will provide relevant data layers for use in exposure estimation, but generally do not provide any method of interpretation or combination of data into metrics meaningful for pesticide exposure estimation.

The proactive development of a set of landscape-level information related to specific crop/climate/exposure regimes, for use by regulatory agencies, academia and research organizations, and the crop protection industry, should be considered for future research efforts. This may include a set of landscape-level data for use in refinement of Step 3 scenarios, as well as additional data suitable for the implementation of catchment level modelling, or to provide input distributions for probabilistic modelling approaches.

The goal of these research efforts will be to provide a reasonable level of confidence for the regulatory community, academia and research organizations, and the crop protection industry, that spatial approaches can be consistently, scientifically, verifiably, and ethically applied to ecological risk assessment. With confidence that the risk assessment process can be enhanced with the appropriate application of spatial technologies, all participants in the risk assessment process will benefit from recent advances in this field.

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2.4 Data layers for integrated spatial analyses

Changes in the environment and the factors influencing those changes are frequently phenomena that differ between places and vary in effect, i.e. demonstrate spatial characteristics. To assess the status of, or model potential changes to, an environmental indicator, those variations should be taken into account. Spatial analysis of environmental indicators requires ancillary data to be available as spatial data layers in a suitable form. The suitability of data is largely determined by its intended use and hence the degree to which it can be integrated with other data.

Suitability is often linked to the scale and projection of the data, but thematic and temporal issues should also be considered. Incoherent thematic classifications between data of different origin in land cover or soil type data sets can complicate integrating the data with other data layers. Similar considerations apply to outdated datasets.

In the following sections a summary of spatial data sets with European coverage is presented. The amount of data available changes with time, but very little freely available data sets have emerged. Progresses in areas of integrating data from different regions to obtain a homogenous European coverage are also rather slow. In principle, data sets with European coverage are available at scale 1:1 million or smaller. Data sets at larger scales do exist, but usually cover smaller areas. Where larger regions are covered, those data sets were included in the following summary. Finally, several ongoing initiatives supporting trans-boundary data compilation are presented to give the reader a sense of the comprehensive data sets projected to be available in the future.

2.4.1 Overview of spatial data

Data for modelling and analysis purposes are available to several varying degrees of detail and spatial coverage. In general, data with continuous spatial coverage at continental scale exist at a scale of 1:1 million (vector data) or 1-km grid size (raster data). Recently, more detailed datasets with European coverage have appeared, such as:

- Global coverage of 3-arc second elevation data (approximately 90m at equator) from the Shuttle Radar Topography Mission (SRTM, USGS, http://srtm.usgs.gov/).

- European coverage of CCM River and Catchment database. The Catchment Characterization and Modelling (CCM) data includes catchments and surface water derived from 250m elevation (and other supporting) data. (JRC, http://agrienv.jrc.it/activities/)
- Global coverage of ortho-rectified historical Landsat TM mosaic at 30m resolution (GeoCover, NASA, http://www.esad.ssc.nasa.gov/).

However, other basic modelling data layers, e.g. soil data, will not be available at scales better than 1:1 million with European coverage for the next few years.

Data at higher resolution are usually not available in form of a harmonized layer covering Europe as a whole. Furthermore, in most cases there is no single provider of the data, with may be the exception of distributors of satellite images. Typically, the tasks of identifying relevant data and sources and integrating such data into a useable format have to be carried out by the user of the data. Annex 1 and Annex 2 of this report are provided to aid the user in identifying specific data layers and providers.

2.4.1.1 Problems of data combination for integrated modelling and analysis

Harmonizing data from different sources to a single coverage is a time-consuming task. Problems to overcome are the application of projection parameters to a common system of geo-referencing and matching common boundaries. More complex is the integration of different thematic data layers. For example, rivers are commonly used as limits of administrative boundaries. The administrative boundaries and the river should therefore coincide in those areas. This demand can be extended to administrative boundaries being defined as in the centre of the river or located along the left or right bank.

When using data of diverse resolution and thematic content the issue of scaling the information becomes important. As an example, the crop cover of a field can be used. At a spatial resolution smaller than the field size a single crop can be attributed to the field. With lower spatial resolution the area covered by a spatial unit would no longer cover a single crop, but include a mixture of different crops. Crop cover is then typically given in form of a percentage of a crop within the spatial unit. The change in attribute assignment is non-trivial and often necessitates a complete change in the data analysis procedure.

2.4.1.2 Sources of detailed national and local data

Sources of detailed data include governmental offices to research institutions and private companies. A non-exhaustive list of national level data sources compiled for the FOCUS process can be found in Annex 1 and Annex 2 of this report (in condensed format) and a more complete version at http://viso.ei.jrc.it/focus.

It is worth noting that for some countries with federal structure, e.g. Germany, the data source depends not only on the thematic content, but also on scale, since large scale data are held and distributed by local rather than federal institutions.

2.4.2 Specific thematic data layers at EU-wide extent

EU-wide data sets that are potentially valuable for use at Step 4 for pesticide exposure assessment are presented in the following sections, including: land cover data, drainage and river data, soils data, meteorological data, and elevation data. Please note that these represent a sample of specific data sets, and availability of data is changing rapidly. Refer to Annex 2 of this report to locate data providers for the most recent data available.

2.4.2.1 Land cover data

The primary source for EU-wide land cover data in Europe is the CORINE (CoORdination of INformation on the Environment) Land Cover (CLC) data set. The CORINE data set provides uniform and comparable land cover data for the territory of the European Union. The CORINE land cover nomenclature is organized on three levels. The first level (5 classes) indicates the major categories of land cover; the second level (15 classes) is for use on scales of 1:500,000 and 1:1,000,000; and the third level (44 classes) is used for projects on a scale of 1:100,000. The minimum mapping unit is 25 hectares.

CORINE Land Cover Classes		
Level 1	Level 2	Level 3
1. Artificial surfaces	1.1. Urban fabric	1.1.1. Continuous urban fabric
		1.1.2. Discontinuous urban fabric
	1.2. Industrial, commercial and transport units	1.2.1. Industrial or commercial units
		1.2.2. Road and rail networks and associated land

		1.2.3. Port areas
		1.2.4. Airports
	1.3. Mine, dump and construction sites	1.3.1. Mineral extraction sites
		1.3.2. Dump sites
		1.3.3. Construction sites
	1.4. Artificial non-agricultural vegetated areas	1.4.1. Green urban areas
		1.4.2. Sport and leisure facilities
2. Agricultural areas	2.1.Arable land	2.1.1. Non-irrigated arable land
		2.1.2. Permanently irrigated land
		2.1.3. Rice fields
	2.2. Permanent crops	2.2.1. Vineyards
		2.2.2. Fruit trees and berry plantations
		2.2.3. Olive groves
	2.3. Pastures	2.3.1. Pastures
	2.4. Heterogeneous agricultural areas	2.4.1. Annual crops associated with permanent crops
		2.4.2. Complex cultivation patterns
		2.4.3. Land principally occupied by agriculture, with significant areas of natural vegetation
		2.4.4. Agro-forestry areas
3. Forests and semi-natural areas	3.1. Forests	3.1.1. Broad-leaved forest
		3.1.2. Coniferous forest
		3.1.3. Mixed forest
	3.2. Shrub and/or herbaceous vegetation association	3.2.1. Natural grassland
		3.2.2. Moors and heathland
		3.2.3. Sclerophyllous vegetation
		3.2.4. Transitional woodland shrub
	3.3. Open spaces with little or no vegetation	3.3.1. Beaches, dunes, and sand plains
		3.3.2. Bare rock
		3.3.3. Sparsely vegetated areas
		3.3.4. Burnt areas
		3.3.5. Glaciers and perpetual snow
4. Wetlands	4.1. Inland wetlands	4.1.1. Inland marshes
		4.1.2. Peatbogs
	4.2. Coastal wetlands	4.2.1. Salt marshes
		4.2.2. Salines
		4.2.3. Intertidal flats
5. Water bodies	5.1 Inland waters	5.1.1 Water courses
		5.1.2 Water bodies
	5.2 Marine waters	5.2.1 Coastal lagoons
		5.2.2 Estuaries
		5.2.3 Sea and Ocean

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The CLC90 data were created by European Topic Centre on Terrestrial Environment Topic Centre of European Environment Agency (http://terrestrial.eionet.eu.int/) and are distributed by the European Environment Agency data service (http://dataservice.eea.eu.int/dataservice). The most recent version (version 2.0) of the CLC90 was updated in December, 2000. An updated version of the CORINE land cover data is currently underway, called the CLC2000, in conjunction with the Image2000 program (see http://terrestrial.eionet.eu.int/CLC2000 and http://image2000.jrc.it/).

The extent of coverage (the EU25), numerous land cover classes, spatial resolution (100m and 250m raster), and temporal resolution (<5 years old) make this an ideal data set for EU-wide and multi-Member State analyses. Source data for the CORINE data set are provided by each member state, and can sometimes be acquired from individual member states in slightly enhanced format (vector instead of raster format, more land cover classes, etc.).

2.4.2.2 Drainage and river data

The location of surface water across the EU is important for a proper examination of potential vulnerability and the subsequent selection of more detailed areas of study, and for proper interpretation of specific landscape results and/or studies. Drainage areas (catchments) are commonly a unit of study utilized for surface water quality issues. For hydrological applications the identification of drainage direction in a river network is essential.

A search for suitable river network data sets revealed that there appears to be very few single EU-wide layers available. National data sets have different copyrights attached, are of varying quality and, in the case of a river network, have different density of the data at the same scale, and are comparatively expensive.

In most cases of EU-wide coverage, the rivers (and catchment areas) have been derived from elevation data. The use of elevation data alone to derive a drainage data set cannot be recommended. The reasons for this are manifold and varied. The resolution of the elevation data may ignore narrow passages, which form an outlet of a basin. The result is a barrier in the data with the possibility of having an artificial outlet in an area of lower elevation elsewhere. Furthermore, in many areas the flow of water has been diverted from its original position with the consequence that the drainage system does no longer follow the elevation data. Other reasons for using elevation data as the only source of information to identify catchments are artefacts caused by the digitization process and an inadequate vertical resolution to identify the flow direction in flat areas.

A more satisfactory approach to the identification of catchments is the integration of elevation data with data containing the actual flow of water. Such information can be derived from a river network data set, and this approach has been applied to some of the presented data sets (e.g., CCM River and Catchment database).

The Catchment Characterization and Modelling (CCM) River and Catchment database (produced by the Agri-Environment Action managed by the Soil and Waste Unit of the JRC Institute for Environment and Sustainability, http://agrienv.jrc.it/activities/catchments/) provides catchment and river segment data based on 250m elevation data across Europe. Under FP6, CCM developed a first version of a European-wide river and catchment database for future use in environmental modeling activities. The database corresponds to a mapping scale of roughly 1:250 000 to 1:500,000, depending on the region.

The European rivers and catchments database (ERICA Version 1998) at scale 1:1,000,000 contains over 1500 catchments to river confluences for the largest rivers in EEA member states. The dataset was developed by EEA to promote analysis using practical hydrological units. The source of the river data set was Collins Bartholomew data at 1:1 million scale. (http://www.bartholomewmaps.com/)

HYDRO1k (http://edcdaac.usgs.gov/gtopo30/hydro/index.asp) is a geographic database developed to provide comprehensive and consistent global coverage of topographically derived data sets, including streams, drainage basins and ancillary layers derived from the USGS' 30 arc-second (approximately 1 km) digital elevation model of the world (GTOPO30, http://edcdaac.usgs.gov/gtopo30/gtopo30.asp)

In some cases, the river network is part of a more extensive set of line data, which increasingly concentrate on the transport infrastructure. While the number of data providers would seem to be extensive, the base data layers seem to originate from few sources. Some of the data sets on offer originate from the same base data.

Digital Chart of the World (DCW)[2] - Data from the DCW is freely available on the Internet. It ties in with the 30” (1km) DEM data available through NIMA and other sources. Other data relate more or less directly to the DCW. GeoComm allows a direct download of some layers

[2] Now: Vector Map Level 0 (VMAP) http://164.214.2.59/publications/vmap0.html

by country from their web site. Global Insight presents a modified data set as a largely improved product and replacement for the DCW.

EuroData (Bartholomew) - The company provides a European multi-layer data set in vector and raster data. The data were used by the ERICA project in the generation of their river basins. It should be pointed out that the product "Euromaps on CD" is not identical to the EuroData layers. Although there is no difference in price, the latter are up-dated first and should be used in preference.

AA Automaps (MapInfo) **-** The road maps contain several separate data layers. The data provider offers the river network layer as a separate product on demand. The data have been used by ADAS, UK and were found to be satisfactory for their purpose.

There are three other providers of combined data sets with European coverage (AND Mapping, ESRI and MapInfo). There seems to be close resemblance between the data sets. ESRI gives as source for their ArcEurope data as the company AND Mapping B.V. According to TeleAtlas the company AND Mapping B.V. is one of their partners. The source of the Cartique® product from MapInfo is not referenced. However, a look at sample data from AND Mapping and Cartique® strongly indicates that both use the same river network data.

2.4.2.3 Soils data

The most commonly used comprehensive soils data set that spans the EU is the European Soil Data Base (v1.0) which consists of a number of databases:

- the Soil Geographical Data Base of Europe at Scale 1:1,000,000 (SGDBE), which is a digitized Eurasian soil map and related attributes,
- the PedoTransfer Rules Data Base (PTRDB) which holds a number of pedotransfer rules which can be applied to the SGDBE,
- the Soil Profile Analytical Data Base of Europe (SPADBE),
- and the Database of Hydraulic Properties of European Soils (HYPRES).

Version 2 extends the coverage to include Eurasia. The data are produced through the Soil & Waste Unit, European Soil Bureau Network, Institute for Environment and Sustainability, Joint Research Centre of the European Commission (http://eusoils.jrc.it/).

2.4.2.4 Meteorological data

One of the most useful meteorological databases is from the Monitoring Agriculture with Remote Sensing (MARS) program (http://mars.jrc.it/). This data set contains historical daily weather observations from several hundred meteorological stations across Europe from 1975 – 2003 (depending on stations). The spatial extent of this data also includes coverage of the new EU member states. The data are interpolated to a 50 x 50 km cell grid structure.

2.4.2.5 Digital elevation data

Elevation data is a fundamental base layer to a wide range of applications, for example delineation of catchments, soil erosion assessment or crop suitability evaluations. Sources of the data are variable; some are derived from satellite images and remotely sensed data, and from digitised topographic maps. At higher resolutions aerial photos and laser are used. There are several data sets containing elevation data with European coverage.

The National Imagery and Mapping Agency of the U.S. (now called National Geospatial-Intelligence Agency, NGA), holds elevation data for most parts of the world at various resolutions, ranging from 30m to 1km. Interesting products are DTED Level 1 and Level 2 having a resolution of 3 arc seconds (approx. 90m), and 1 arc second (approx. 30m), respectively. Access to the 1 arc second data is, however, restricted. For continental applications the "Global 30-Arc-Second Elevation Data Set", referred to as GTOPO30, is also suitable[3]. The data for Europe originates from the "Digital Terrain Elevation Data" (DTED Level 0) and the "Digital Chart of the World" (DCW).

The Shuttle Radar Topography Mission (SRTM) is a joint NASA-NGA (National Geospatial-Intelligence Agency) partnership. USGS EROS Data Center (EDC) distributes and archives SRTM data for NASA in accordance with policy guidelines set forth by NASA-NGA (http://srtm.usgs.gov/). The non-US SRTM data are 3 arc second (90m) resolution data and will be available for online download.

The EuroGeographics organisation was formed by a merger of CERCO (Comité Européen des Responsables de la Cartographie Officielle) and MEGRIN (Multi-purpose European Ground Related Information Network) 2000. The web-site[4] contains a list of data layers to be

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[3] http://edcwww.cr.usgs.gov/Webglis/glisbin/guide.pl/glis/hyper/guide/gtopo_30

[4] http://www.eurogeographics.org/

developed at scale 1:1million (EuroGlobalMap) and at scale 1:250,000 (EuroRegionalMap). While the availability of DEM data through EuroGeographics is not clear, the site is useful through its links to National Mapping Agencies.

An extensive catalogue of DEM data was compiled by B. Gittings, University of Edingburgh, U.K.[5] The catalogue concentrates on data rather than a specific category of providers. Yet, data from private companies are not well presented. Furthermore, the list was last up-dated in January 1997. Several entries have changed since then, with data being no longer available or being held by different institutions.

National Mapping Agencies offer very diverse products at largely different prices. For example, the only official DEM for Ireland uses 1km grid spacing. DEMs for some countries can only be obtained through agencies of military installations or with written consent of the Ministry of Defence (e.g. Finland, Greece, Italy, NIMA DTED Level 1).

A compilation of providers of DEM data is given in Appendix A2. The list contains private companies, public institutes and national administrations, depending on the country and, in some cases, the region of a country. It should be noted that only those prices were included in the table, which are publicly available from product descriptions. In some cases, discounts for volume data orders are available.

2.4.3 Multi-layer data sets

Several general datasets are also available at the EU-wide level. These data sets contain multiple layers, each representing various 'themes'.

Bartholomew[6] publish vector and raster maps at various scales, ranging from street maps to global data sets. Data can be provided in various map data formats or projections for input into geographical information systems, desktop mapping and other applications. (see also http://www.graticule.com/MapData/Bartholomew.htm)

An interesting data set with pan-European coverage is the data set "Europe 1:1,000,000 Data". The data set covers the whole of Europe from the Atlantic Ocean to the Caspian Sea

[5] Digital Elevation Data Catalogue: http://www.geo.ed.ac.uk/home/ded.html

[6] http://www.bartholomewmaps.com/index.html

and from the Mediterranean Sea to the Arctic Ocean. It includes all of the European Union (EC), western Russia, Iceland, the Canary Islands and the Azores.

Included are administration boundary changes, a fully indexed gazetteers of over 77,000 towns and 3,000 mountain peaks, and an overview layer of country boundaries (from the Bartholomew 1:20M database). Some of the thematic data layers of interest include: administrative layer, contours and bathymetry, drainage: permanent and impermanent, roads, major built-up-areas, woodland, water: lake, lagoon, marsh, glacier, etc., lines with river names.

The Digital Chart of the World (DCW) is an Environmental Systems Research Institute, Inc. (ESRI) product originally developed for the US Defence Mapping Agency (DMA) using DMA data. The DCW 1993 version at 1:1,000,000 scale was used. The DMA data sources are aeronautical charts, which emphasize landmarks important from flying altitudes. Note that the completeness of the thematic categories present in each layer will vary. Some of the thematic data layers of interest include: populated places, roads, drainage, land cover, ocean features, physiography, transportation, and vegetation.

Vector Map (VMap) Level 0 is an updated and improved version of the National Imagery and Mapping Agency's (NIMA) Digital Chart of the World (DCW®). The VMap Level 0 database provides worldwide coverage of vector-based geospatial data, which can be viewed at 1:1,000,000 scale. It consists of geographic, attribute, and textual data stored on compact disc read-only memory (CD-ROM). The primary source for the database is the 1:1,000,000 scale Operational Navigation Chart (ONC) series co-produced by the military mapping authorities of Australia, Canada, United Kingdom, and the United States (from GeoCommunity)[7]. The thematic content of the VMAP0 data set is very similar to the one of the DCW.

EuroGeographics is the association of the European National Mapping Agencies, with 40 members from 38 countries. The aim of the association is to achieve interoperability of European mapping (and other GI) data within 10 years". (http://www.eurogeographics.org/AboutUs/index.htm)

The following pan-European datasets are planned or currently available:

[7] http://store.geocomm.com/viewproduct.phtml?catid=25&productid=1194

- SABE: Seamless administrative boundaries dataset on the scale 1: 100,000 and 1: 1,000,000, with administrative units down to NUTS 5 available (e.g, commune, municipios etc).

- EuroGlobalMap: A 1:1 million topographic dataset that is the European contribution to the Global Map project. The database contains the following six themes: Administrative Boundary, Hydrography, Transportation, Built-up Areas, Elevation, Named Location.

- EuroRegionalMap **:** A 1:250,000 scale topographic dataset. The database will contain the following six themes: Administrative Boundary, Hydrography, Transportation, Built-up Areas, Elevation, Named Location.

GISCO is the Geographic Information System for the European Commission. Originally conceived as a prototype GIS cell that would serve a wide spectrum of users and uses, the GISCO project has developed a service-oriented dimension, namely in geographical database development, thematic mapping, desktop mapping and dissemination of data. The data set contains 6 groups of basic topographic layers and 7 thematic layer groups. Most data are available at scale 1:1 million or smaller. Basic topographic data includes: Administrative Data, Hydrography, Altimetry, Infrastructure, and Support. Thematic data include: Community Support Frameworks, Environment, Industrial Themes, Infra Regional Statistics, Land Resources, Nature Resources, and World Data.

The United Nations Environment Programme GRID database is maintained for the purpose of assisting the international community and individual nations in making sound decisions related to resource management and environmental planning, and where applicable providing data for scientific studies. Within the overall GRID-network, GRID-Geneva focuses on the acquisition or creation, documentation, archive and dissemination of Global and European digital geo-referenced environmental data. (http://www.grid.unep.ch/data/index.php). GRID thematic data includes: Atmosphere, Biodiversity, Boundaries, Climate, Ecological/Life zones, Human related, Hydrology, Land Cover, Oceans & Seas, Physical Geography, Soils, and Vegetation index.

2.4.4 Initiatives supporting trans-boundary data compilation

While there are numerous organizations and private companies collecting and distributing data there are also initiatives, which do not compile data directly, but support such efforts at national and international level. The principal initiatives related to the collection and provision of spatial data for landscape analysis are briefly presented hereafter. The purpose is not to provide an exhaustive project description and inventory, but to simply give the reader a starting point in which to investigate these initiatives further.

Current initiatives described include:

- INSPIRE (Infrastructure for Spatial Information in Europe)
- Water Framework Directive
- EUFRAM (Probabilistic approaches for assessing environmental risks of pesticides)
- SAGE (Service for the Provision of Advanced Geo-Information on Environmental Pressure and State)
- GINIE (Geographic Information Network In Europe)
- Agri-Environment Action
- Catalogue of European Spatial Datasets

2.4.4.1 INSPIRE[8] (Infrastructure for Spatial Information in Europe)

INSPIRE is a recent initiative launched by the European Commission and developed in collaboration with Member States of the European Union and accession countries. It aims at making available relevant, harmonised and quality geographic information to support formulation, implementation, monitoring and evaluation of Community policies with a territorial dimension or impact.

The INSPIRE initiative intends to trigger the creation of a European spatial information infrastructure that delivers to the users integrated spatial information services. These services should allow the users to identify and access spatial or geographical information from a wide range of sources, from the local level to the global level, in an inter-operable way for a variety of uses. The target users of INSPIRE include policy-makers, planners and managers at European, national and local level and the citizens and their organisations. Possible services are the visualisation of information layers, overlay of information from different sources, spatial and temporal analysis, etc.

[8] http://inspire.jrc.it/home.html

The spatial information infrastructure addresses both technical and non-technical issues, ranging from technical standards and protocols, organisational issues, data policy issues including data access policy and the creation and maintenance of geographical information for a wide range of themes, starting with the environmental sector.

2.4.4.2 Water Framework Directive[9]

The objective of the Water Framework Directive (WFD) is to establish a Community framework for the protection of inland surface waters, transitional waters, coastal waters and groundwater, in order to prevent and reduce pollution, promote sustainable water use, protect the aquatic environment, improve the status of aquatic ecosystems and mitigate the effects of floods and droughts.

Under this Directive, Member States identified all the river basins lying within their national territory and assigned them to individual river basin districts. River basins covering the territory of more than one Member State were assigned to an international river basin district and a competent authority was designated for each of the river basin districts.

At the latest, four years after the date of entry into force of this directive, Member States must complete an analysis of the characteristics of each river basin district, a review of the impact of human activity on the water, an economic analysis of water use and a register of areas requiring special protection. All bodies of water used for the abstraction of water intended for human consumption providing more than 10 m^3 a day as an average or serving more than 50 persons must be identified. The described characterisation is due at the end of 2004.

EUFRAM[10] (Probabilistic approaches for assessing environmental risks of pesticides)

The main work of the EUFRAM project will be done by a core partnership of 27 organisations from government, industry and academia, and comprises three main parts.

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[9] Directive 2000/60/EC of the European Parliament and of the Council of 23 October 2000, establishing a framework for Community action in the field of water policy [Official Journal L 327, 22.12.2001]. http://europa.eu.int/scadplus/leg/en/lvb/l28002b.htm

[10] http://www.eufram.com/index.cfm

- Development of a draft framework of basic guidance for risk assessors. The topics to be addressed include:
    - role and outputs of probabilistic assessments
    - methods of uncertainty analysis
    - probabilistic methods for small datasets
    - how to report and communicate results
    - how to validate probabilistic methods
    - how to improve access to existing data
    - requirements for probabilistic software and databases.

  The framework will also include case studies of probabilistic risk assessment, showing how the methods can be applied to assessing impacts of pesticides on terrestrial and aquatic organisms. The first draft of the framework will be published at the end of 2004.

- End-user testing. In 2005-2006, the draft framework will be subjected to extensive testing and refinement. A series of three workshops will be organised for potential users, who will be encouraged to trial the framework in their own organisations. Feedback from the users will be used to refine the framework, and it is intended that the final version will be suitable for adoption as standard guidance at the European level.

- Public network. At the start of the project a public network will be established to share information about research needs, ongoing projects and future activities related to the continuing development of probabilistic methods for pesticides. This will help to prioritise outstanding research needs, and will encourage initiatives aimed at addressing those needs. The network will also be used to disseminate progress reports and outputs from EUFRAM itself. The public network is open to everyone.

2.4.4.3 SAGE[11] (Service for the Provision of Advanced Geo-Information on Environmental Pressure and State)

The "Service for the Provision of Advanced Geo-Information on Environmental Pressure and State" (SAGE) offers a comprehensive product portfolio to serve the demands coming from the European Water Framework Directive (WFD) and the upcoming regulations of the Thematic Strategy on Soil Protection (usually referred to as Soil Thematic Strategy (STS)).

Public and private partners together have established core services addressing basic geo-information needs of the environmental community. They serve as the basis of customised end-user applications supporting the national and local implementation of the WFD and STS.

All SAGE products have been approved as fulfilling the monitoring requirements of all European partner agencies, labelled as proven and sound by independent scientific reviewers, and designed for efficient implementation by the service provider team.

As the SAGE services are designed in an open and modular way, many other European environmental and planning authorities will be able to profit from the SAGE products as well.

(Description taken from web site at http://www.gmes-sage.info/)

2.4.4.4 GINIE[12] Geographic Information Network In Europe

GINIE is a research project funded by the Information Society Technology Programme of the EU for the period November 2001- January 2004. Its partners are EUROGI, the European Umbrella Organisation for Geographic Information, the Open GIS Consortium Europe representing the Geographic Information (GI) industry, the Joint Research Centre of the European Commission, and the University of Sheffield (Coordinator).

The aim of the project was to develop a deeper understanding of the key issues and actors affecting the wider use of GI in Europe, and articulate a strategy to promote such wider use that is consistent with major policy and technological developments at the European and international level. Close attention has been paid to the role of GI in supporting European policies with a strong spatial impact (agriculture, regional policy, transport, environment), e-

[11] http://www.gmes-sage.info/

[12] http://www.ec-gis.org/ginie/

government, the re-use of Public Sector Information, and the recent initiative to develop an Infrastructure for Spatial Information in Europe (INSPIRE).

To achieve its objectives, the project has organised a series of specialist workshops, commissioned analytical studies, collected numerous case-studies of GI in action, and disseminated widely its findings across Europe and beyond. Through its activities GINIE has involved more than 150 senior representatives from industry, research, and government in 32 countries, and contributed in building the knowledge necessary for an evidence-based geographic information policy in Europe. The project presented its findings to a high-level audience of senior decision makers in government, research and industry at its final conference in Brussels in November 2003. Full details and all the project reports are available in the documents section of this site.

(Description taken from web site at http://www.ec-gis.org/ginie/)

2.4.4.5 Agri-Environment Action[13]

Members of the the Agri-Environment Action work on the following issues:

- Integration of spatial information layers at different scales for the estimation of land cover change in rural areas. The work consists of the methodological development of tools for the implementation of a sustainable EU agricultural policy.
- Monitoring and modeling of European landscapes, including the test of selected pressure indicators over European landscapes.
- Further development of a European river and catchment database (CCM) at intermediate scale (1:250,000 to 1:500,000) in support to environmental reporting activities of DG Environment and EEA.
- Making available JRC’s expertise and competence for understanding the linkages between agriculture and environment, with particular emphasis on the spatial component.

(Description from web site at http://agrienv.jrc.it/activities/)

[13] http://agrienv.jrc.it/activities/

2.4.4.6 Catalogue of European Spatial Datasets[14]

In order to facilitate a more effective accessibility of European spatial datasets, an assessment was carried out by the GeoDesk of the WUR to identify and describe key datasets that will be relevant for research carried out within WUR and MNP. The outline of the Metadata catalogue European spatial datasets, the classification of the datasets and the use of specific standards, is based on the work which was be done by the INSPIRE (INfrastructure for SPatial InfoRmation in Europe) initiative. The objective of the report is that it can speed up the process for identification of suitable datasets during the following steps: - to inform on the existence of European spatial datasets that could be relevant for a specific project; - to evaluate if a dataset will be suitable by exploring the metadata; - to indicate if a relevant spatial dataset is available and give directions how it can be obtained. (Abstract of report, Willemen et al., 2004).

2.4.5 Identification of data gaps

The discussion of the suitability of spatial data and demands on data properties are strongly related to the particular use of the data within an assessment, and the precision and scale of the conclusions which are to be drawn from the final results. Some fundamental theoretical investigations are available about the dependence of precision, resolution, accuracy of data bases and the accuracy of the results after using the data in projects of considerable complexity (e.g. using various datasets, simulation models, conservative assumptions, etc.), yet applied guidance for particular practical situations is hardly available.

As a principle rule of thumb, the resolution and accuracy of the data should fit into the overall accuracy and scale of the approach (or into that of sub-processing steps of more complex evaluations respectively), comprising e.g. various data sources, substance properties, product use data, model and submodel assumptions, and the conclusions drawn (e.g. mitigation measures).

Conservative (‘realistic worst case’) assumptions can replace a lack of actual information (data) and hence should guarantee the overall conservative character of the estimated exposure (e.g. use of spray drift deposition values measured on turf (or bare soil) not on water body surfaces, a water body depth of only 30 cm which represents a potential habitat

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[14] http://www.alterra.wur.nl/

for the most sensitive species (e.g. fish), all farmers spraying at the same time within a landscape, wind blows toward the water body, no intervening vegetation, etc.).

Looking at the spray drift entry route of plant protection products into water bodies, the distance (edge-of-the-field to water body) is a major driving landscape factor of potential exposure of aquatic habitats. As the potential drift deposition significantly decreases over the first 30 metres from the treated field, high-resolution measurements in the landscape are necessary to reasonably calculate the potential drift deposition with distance. Such spatial databases, describing the high-resolution field-to-water body distances are generally not available. For example, examining a 1:25,000 scale dataset on land use/cover (e.g. the ATKIS DLM/25 dataset of Germany), the given land use polygons do frequently directly border on the water body edges (i.e. distance = 0m). This is not the case in reality as evaluated by different authors using aerial imagery or field observations. Using this database without any validation would lead to strong underestimations for distances and would lead to drift deposition values greater than those in reality.

More realistic measurements for the distances can be derived using high-resolution remote sensing data (e.g. aerial imagery, IKONOS, RapidEye, etc.). From this example it can be concluded that ready-to-use spatial datasets are not available to quantify an important landscape factor covering significant regions in EU, however, the required real-world information can be *derived* using readily available, up-to-date data (e.g. high resolution remote sensing data). An analogous conclusion can be drawn for the identification of intervening vegetation reducing spray drift or runoff. Therefore, in the discussed example the principle demand for distances and land cover derived from high-resolution data leads to the demand for an increase of the operability to derive relevant landscape information from high resolution and up-to-date remote sensing imagery.

Local environmental conditions affecting potential entries of plant protection products into water bodies due to runoff are discussed in this report. Correspondingly, local data on the land use/cover, weather, soil properties, composition of buffers, slope, etc. are principally required in high resolution and up-to-date, in order to improve Step 3 edge-of-the-field modelling. For the majority of the investigations the perfect combination of datasets will not be available. Nevertheless, using alternative ways to derive the necessary data or by making reasonable conservative assumptions for data which are currently not available (‘data gap’), more realistic input for Step 3 scenarios can be derived for specific questions using the currently available databases (e.g. databases available on Member State level).

The local or regional data on the presence, type, and constitution of drainages demonstrates a clear lack of information for refined Step 3 drainage calculations. Although the detailed data is not readily available, reasonable assumptions using available geodatabases allow approaches the get more realism into Step 3 model scenarios: land use information, data on the presence of water bodies, weather data and soil property data can be combined to conservatively conclude scenario improvements, and to derive more realistic assessments on potential entries of plant protection products into water bodies due to drainage.

A considerable data gap on local environmental conditions is given for characterisation of the water bodies. Currently available databases on Member State level are mainly derived from topographic maps and do not contain the spectrum of properties which would significantly increase the prediction of local PECsw, e.g., due to dilution and dispersion, or regarding the consideration of environmental fate of the substances in the aquatic system. Although in a few of the available geodata bases some categorisation is made about the size or type of a water body, principle gaps of information are about the water body width, depth (volume with time), flow, sediment, bank, substrate, water plants, ecological description, etc. The development of these data could significantly increase the realism of PECsw/sed, and hence the aquatic risk assessment. Although the demand for the mentioned information on water bodies is obvious, even better data and characterisations should fit in the overall precision and goal of the exposure and risk assessment, as well as risk management and authorisation: The risk assessment principally aims at protecting non-target populations of entire PPP use regions and the authorisation has to be valid for large regions and valid over long periods of time. Therefore, exaggeration of the demands on (thematic) precision and accuracy of local data in space and time does not necessarily enhance the overall basis for decision making. It is not the data that is needed to predict a local PECsw for “Thursday afternoon, 19. May”, instead, it is the data which allows a locally realistic but still conservative, and most of all robust, estimation of potential exposures of aquatic habitats, which results in a realistic worst case evaluation for entire use regions over long periods. Hence, an “intelligent simplicity” should guide the demand for resolution, precision, and accuracy of the data.

In case more detailed data on the aquatic systems become available, refined characterisations of the environmental fate properties of substances in real aquatic systems can be applied. At present, experiments (e.g. photolysis, degradation, adsorption, etc.) are principally performed in the lab, using artificial systems, which aim more at a principle demonstration of fate routes and dissipation, than in putting these properties in context to natural aquatic systems (e.g., using simulation models).

To summarize, it can be stated that harmonised and up-to-date ready-to-use key databases of appropriate resolution and accuracy are still to be developed. An overview is given in Table 2.4-2 below, which tries to put potential data gaps in context. The demand for ideal geodata should be taken as a challenge for further developments of databases and data processing methods, and should not be read inversely (in other words, saying that Step3 improvements using geodata should not wait for the final perfect data point).

Table 2.4-2: Overview of potential data gaps and alternative ways to derive relevant information for refined step3 model calculations

Landscape Factor	Data base	Ready-to-use availability	Alternative ways to derive appropriate information	Remark
crop – water body distance	high resolution large scale land use, land cover data; hydrology	low	high	high resolution remote sensing data available and affordable; considerable processing cost due to low automation
composition of buffer with respect to functional properties to reduce spray drift or runoff	high resolution large scale land use, land cover data; hydrology	low	high	high resolution remote sensing data available and affordable; considerable processing cost due to low automation
soil properties	medium scale soil database on soil properties, differentiated by land use	medium	low	most relevant for runoff and drainage calculations; only available at Member State level at best
water body width	hydrology	medium	low	standard topograpic database often includes categorised width classes
water body depth (t)	hydrology	low	low	due to the temporal character of the water volume demands should not be exaggerated
water body physico-chemical properties	hydrology	low	low	important e.g., for actual environmental fate of substances and ecological charactersation
water body sediment layer	hydrology	low	low	important e.g., for actual environmental fate of substances and ecological charactersation
water body plants	hydrology	low	low	spray drift deposition, runoff, environmental fate, ecology
water body geometry	hydrology	low	low	PECsw, environmental fate
drainage	drainage map	low	medium	local information on presence, type, constitution of drainage pipes
ecological data, habitat quality	local data on actual ecological constitution or potential habitat quality	low	low	presence of species, recovery, migration
slope	elevation, slope	medium	medium	potential runoff entry; need detailed elevation data for local slope calculation

From the table above some points for further research can be easily derived, e.g. in the field of reasonably improving the characterisation of water bodies, the development of GIS- and Remote Sensing methods to make up-to-date data available (and affordable), and on the better fit of environmental fate studies with real natural scenarios.

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3 REVIEW OF ECOLOGICAL CONSIDERATIONS AT THE LANDSCAPE LEVEL

Introduction

This chapter provides an overview of the current state-of-the-art concerning the potential implementation of ecological approaches to aquatic risk assessment of plant protection products (PPPs) at the landscape level. Although higher-tier effects assessment in the EU has developed significantly over the last decade, recent changes in surface water exposure assessment resulting from the implementation of the FOCUS scenarios mean that it is timely to review the integration of the effects and exposure assessment at the higher tier.

In the first section of the chapter (3.1), an overview is provided of the current approach to aquatic risk assessments under 91/414/EEC. This discusses protection aims and the legislative background, describes the current risk assessment process, and reviews the implications of the recent changes that have been made to surface water exposure scenarios via FOCUS.

At present, the ecological characteristics (abiotic and biotic) of the surface water scenarios are not well described. The development of this information could potentially be used in the future to refine both the exposure and effects assessment. Existing data and tools for such an approach are briefly reviewed in Section 3.2. How these factors may influence the effects and exposure assessments is then discussed in Sections 3.3 and 3.4 respectively.

Finally, moving to the landscape level provides opportunities for considering recovery potential, both internally (from within the water body of concern) and externally (from neighbouring waters). Potential approaches for developing these techniques have been reviewed in Section 3.5.

3.1 Legislative Background and Protection Aims

3.1.1 In the context of Directive 91/414/EC (EU, 1991)

Whilst Directive 91/414/EEC provides for the protection against unacceptable effects from pesticides on the aquatic environment, it does not specify to the full extent what constitutes an unacceptable influence. The Guidance Document on Aquatic Ecotoxicology (Document SANCO/3268/2001) notes that the sustainability of populations of non-target organisms should be ensured, and that structural and functional endpoints should be regarded of equal importance.

Some discussion of acceptability of effects is provided by Brock & Ratte in the CLASSIC guidance document (Giddings *et al.*, 2002). The criteria proposed for acceptability of effects are summarised in Table 3.1 below

Table 3.1 Criteria for acceptability of effects to non-target aquatic organisms (Giddings *et al.*, 2002)

<u>No decrease in biodiversity.</u> This concerns negative effects on: Overall species richness and densities (expressed e.g., as the number of taxa, diversity indices -or scores of multivariate techniques- for the total community or for taxonomic or functional groups). Population densities of ecological key species (i.e. species that play a major role in ecosystem performance, productivity, stability, resilience), e.g., - species that are critical determinants in trophic cascades (e.g. piscivorous fish; large cladocerans); - species which are “ecological engineers” i.e., those that have a large influence on the physical properties of habitats (e.g. rooted submerged macrophytes). Population densities of indicator species - species with a high “information” level for monitoring purposes; - species protected by law and regionally rare or endangered species). <u>No impact on ecosystem functioning and functionality</u> This concerns negative effects on: Water quality parameters (e.g. increase of toxic algae; oxygen depletion); Harvestable resources (e.g. fish); <u>No decrease in perceived aesthetic value or appearance of the water body such as:</u> Disappearance of species with a popular appeal (e.g. dragonflies; water lilies). Visual mortality of individuals of fish, frogs, water fowl and other vertebrates. Symptoms of eutrophication (e.g. algal blooms).

3.1.2 In the context of Directive 2000/60/EC (EU, 2000)

The Water Framework Directive (2000/60/EC) aims to achieve “good status” for European surface water. “Good status” is determined by both ecological and chemical criteria (Table 3.2). Good ecological status implies that the water body meets both biological and physico-chemical criteria to permit the long-term viability of aquatic organisms. For plant protection products, the aim is to produce chemical quality standards which will permit good ecological status to be achieved both from peaks of exposure (maximum acceptable concentrations) and from longer-term exposure (environmental quality standards). This approach to ecological status is thus broadly consistent with the requirements under Directive 91/414/EC since it refers to maintaining both structure and function of aquatic ecosystems.

Table 3.2 Biological and physicochemical criteria for a "good status" in the context of the Water Framework Directive

The values of the biological quality elements (i.e. composition and abundance of aquatic flora including phytoplankton, benthic invertebrate fauna and the composition, abundance and age structure of fish fauna) show slight deviation from reference conditions, thus meaning low levels of distortion resulting from human activity.

The levels of the general physico-chemical quality elements (i.e. oxygen concentration, temperature, acidity, salinity) do not exceed the range ensuring ecosystem functioning and the achievement of the values associated to biological quality elements at good status.

The concentrations of specific synthetic and non-synthetic pollutants should not be in excess of the standards set in accordance with the procedure detailed in section 1.2.6 of the directive without prejudice to Directive 91/414/EC (EU, 1991) and Directive 98/8/EC (<EQS) (EU, 1998).

The biological status of a water body is assessed through the comparison with a reference biological status in an ecological quality ratio (EQR) 15. The reference biological status may be either determined through monitoring studies or predicted from modelling based on hydro-morphological criteria.

3.1.3 Definition of Water Bodies for Protection

The broad aims of Directives 91/414/EC (EU, 1991) and 2000/60/EC (EU, 2000) are consistent in that both aim to ensure the long-term viability of aquatic ecosystems. These

[15] Guidance on establishing reference conditions and ecological status class boundaries for inland surface waters. Produced by Working group 2.3-Reference conditions for inland surface waters (REFCOND), Final version, 30 April 2003.

requirements are provided with no distinction among water bodies i.e. without taking the type (e.g. static or running water), location in the landscape, relative sensitivity of the aquatic system (e.g. resilience capacity) or even "economic status" of the water body into consideration. However, at present there is no legislative instrument that allows these differences to be taken into account. Even if this were so, transposition to the field would require important practical developments.

As far as the sensitivity of aquatic ecosystem is concerned, a reasonable approach would be to develop different levels of protection according to the resilience of the water body under consideration. However, this implies that levels of protection are defined considering ecological aspects at the scale of the water body but also considering its connectivity at the scale of the landscape, which may finally change from an ecological case to another.

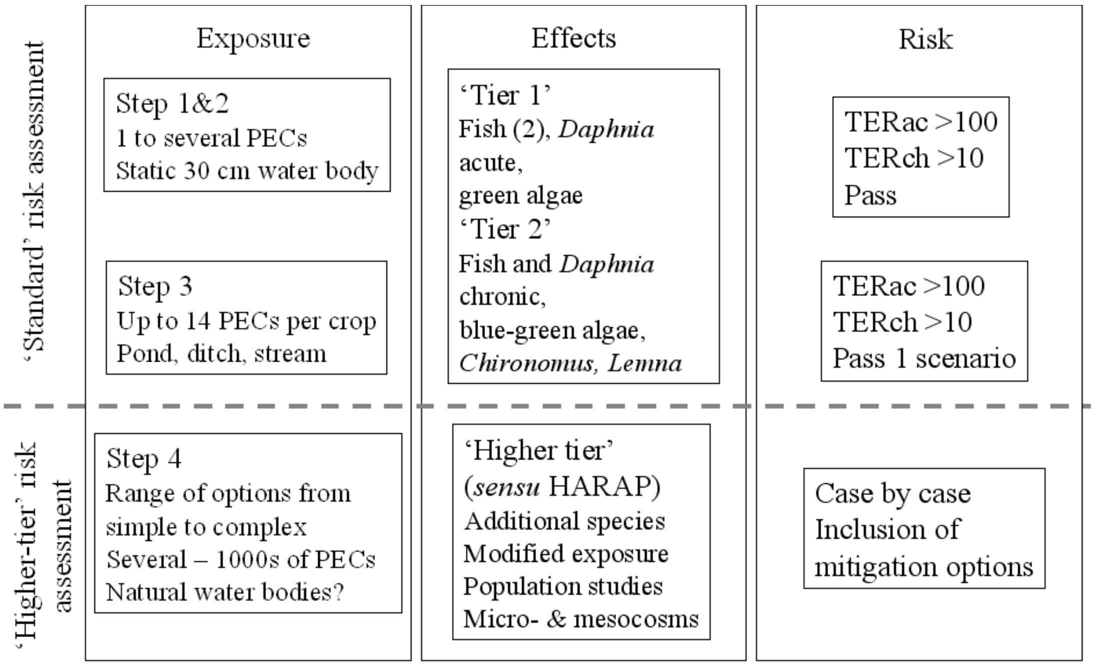
Subsequently, all surface water is treated similarly.

3.1.4 Current risk assessment process

Directive 91/414/EC (EU, 1991) requires that “risks of unacceptable effects for the environment are assessed” before any authorisation of a product is granted. It is further stated that *“since the evaluation is based on a limited number of representative species, it shall be ensured that use of the plant protection product does not have any long term repercussions for the abundance and diversity of non-target species”*.

In order to meet these requirements, risk assessment is commonly based on “worst case” ecotoxicity and exposure assumptions coupled with the use of uncertainty factors. The ecotoxicological profile of a plant protection product is determined from the studies that are required in Annexes II and III of the Directive. A brief description of the risk assessment process under 91/414/EC is described below and the process is summarized in Figure 3.1.

Figure 3.1 Overview of the Aquatic Risk Assessment Process Subsequent to the Recommendations of the FOCUS surface water scenarios report and EU Aquatic Ecotoxicology Guidance Document (SANCO 3268 rev. 3, 2002). ***NB In practice, results from higher-tier effects assessments could be compared to Step 3 calculations, and similarly results from Step 4 exposure calculations could also be compared to lower-tier effects assessments***



3.1.4.1 Effects Assessment

The standard ecotoxicity package generated for PPPs includes *Daphnia* acute (48 h median effective concentration EC50) and chronic (21 d no observed effect concentration NOEC), two fish acutes including a coldwater and warmwater species (96 h median lethal concentration LC50), fish chronic (28 – c. 60 d NOECs) and algal toxicity tests (72-120 h EC50s). The exceptions to this are:

- for compounds which dissipate very rapidly (median dissipation time, DT50, in the water phase in water-sediment systems <2 d), where chronic *Daphnia* and fish studies are not required;
- for herbicides where tests with a second algal species and *Lemna* are also required;
- for insecticides where testing on *Chironomus* may be required.

Tests are often required with the formulated product and under special circumstances they may also be required for metabolites.

These studies aim:

- to provide toxicological thresholds for organisms representing sensitive species from three trophic levels, namely predators (fish), secondary consumers (invertebrate species) and primary consumers (algae[16]). Thresholds for these species are also used in the classification process;
- to identify the most sensitive group of organisms, on which the risk assessment for aquatic ecosystems should be based.

3.1.4.2 Risk Assessment

The maximum predicted exposure concentration (PECmax) in the edge of field water body derived from the FOCUS surface water models is used for comparison to the acute toxicity data. If appropriate to the mode of action (i.e., so long as effects do not occur close to the onset of exposure as for example with some respiratory inhibitors), time-weighted average concentrations (PECtwa) of a duration appropriate to the test duration are calculated for comparison to chronic exposure concentrations. The dissipation measured in laboratory water-sediment fate studies is typically used to parameterise the calculations of time-weighted averages.

In the acute risk assessment, toxicity values (LC/EC50s) are divided by the PECmax to derive an acute toxicity exposure ratio (TERac) and chronic NOECs are divided by the PECtwa (if appropriate) to derive a chronic toxicity exposure ratio (TERch). If the TER is <100 for acute fauna studies, or <10 for chronic fauna studies and studies on plants, further evaluation of potential risks is triggered. In essence this means that uncertainty factors of 100 (for acute endpoints) and 10 (for chronic endpoints) are applied to the standard data package.

Under 91/414/EEC, algal studies are considered to be chronic studies because they include many generations of the test organism, and the EC50 is used as the effect endpoint. This approach has recently been validated in a detailed review by Brock et al. (2000a), who compared the results of mesocosm studies on herbicides with predictions of effects based on laboratory toxicity data (including the uncertainty factor of 10). Brock et al. found that the

[16] and the in case of herbicides, macrophytes

laboratory risk assessment criteria under 91/414/EEC (i.e., use of EC50 and an uncertainty factor of 10) provided sufficient protection of effects observed in mesocosms from the uses of herbicides. A similar validation of the Tier 1 assessment scheme for insecticides has also demonstrated the robustness of the approach (Brock et al., 2000b).

3.1.4.3 Higher-tier Assessment under 91/414/EEC

There is a range of possibilities for refining the effects assessment if evaluations indicate that there are potential concerns. These are described in the HARAP (Campbell et al., 2000) and CLASSIC (Giddings et al., 2002) guidance documents. Further definition of the implementation of higher-tier studies into the PPP evaluation process is also included in The Guidance Document on Aquatic Ecotoxicology (SANCO/3268, 2002). The range of higher-tier options included in these documents is summarised briefly below.

<u>Single species studies</u>

The testing of additional species from taxonomic groups identified as being of potential concern may allow the reduction of the uncertainty factor applied in the preliminary assessment by up to a factor of 10 (i.e., the acute TER trigger may be reduced from 100 to a minimum of 10), recognising that species sensitivity is a key uncertainty in the preliminary assessment and the availability of additional species data reduces this uncertainty.

Species sensitivity distribution (SSD) approaches are also gaining wider acceptance as a method of refining the effects endpoints. By ranking toxicity values and plotting them as cumulative percent rank versus LC/EC50 on a log-normal plot, the distribution of sensitivities of organisms can be described. This distribution can then be used to predict a concentration which will have low effects (typically the lower 10th or 5th percentile).

Modified exposure studies (which mimic the dissipation of the compound, rather than maintaining exposure concentrations as in standard studies) may be conducted if studies have identified that a particular group of organisms is at risk, and dissipation data suggest that standard laboratory studies may overestimate potential effects under field conditions. In these studies, exposure is modified either by the addition of sediment or by simulating the dissipation profile of the compound using a variable flow dosing rig. The effects endpoints derived (usually demonstrating lower toxicity than the standard maintained exposure studies) may then be used in place of the endpoints generated in standard tests in the assessment.

Multi-species studies

Where the risk assessment indicates potential concerns, populations and communities of organisms may also be tested under conditions more relevant to the field in micro- and mesocosm studies. There has been extensive international discussion and guidance generated for such studies with PPPs over recent years (see HARAP and CLASSIC).

The Guidance Document on Aquatic Ecotoxicity (SANCO/3268, 2002) makes recommendations about how such studies should be reported. For the relevant taxonomic groups in the study, a no observed effect concentration at the community level (NOECcommunity) is derived using appropriate statistical techniques (e.g., Principal Response Curves). In addition, NOECs for populations of relevant organisms are reported (NOECpopulation). Where there are effects at the community or population level, the time taken for recovery to occur is reported.

The NOECcommunity, the NOECpopulation and the time taken for recovery are then used to determine a no observed ecologically adverse effect concentration (NOEAEC). The NOEAEC is defined as the concentration at or below which no long-lasting, adverse effects were observed in a particular higher-tier study (e.g. mesocosm). No long-lasting effects are defined as those effects on individuals that have no or only transient effects on populations and communities, and are considered of minor ecological relevance (e.g., effects that are shown not to have long-term effects on population growth, taking into account the life-history characteristics of the organisms concerned). Different recovery rates may therefore be acceptable for different types of organisms. As preliminary guidance (but depending on the study design and life-history of the organism concerned), recovery within a period of 8 weeks is considered acceptable for defining the NOEAEC (i.e., if initial effects are observed, but recovery is observed within 8 weeks, then the treatment concentration can be considered to have no ecologically adverse effects). The NOEAEC can therefore be higher than the NOECcommunity or NOECpopulation since recovery is incorporated.

If the study is considered to be robust and relevant for the concerns identified, the NOEAEC may be used for a direct comparison with the relevant edge-of-field PEC. Otherwise, it is suggested that an appropriate uncertainty factor should be applied. At present, there is no precise guidance on what uncertainty factors are appropriate, and these are generally applied on a case-by-case basis, taking into account the nature of the concern identified and the available data.

The NOEAEC (in combination with a uncertainty factor, if appropriate) can then be used to define an Ecologically Acceptable Concentration (EAC). The EAC was defined at the HARAP workshop "…..as being the concentration at or below which no ecologically adverse effects would be expected. Depending on the type of study, this can be defined either directly (e.g. from semi-realistic multi-species or field studies) or through the application of appropriate uncertainty factors (e.g., with additional single-species tests). Expert judgement is needed in the derivation of an EAC".

So while the NOEAEC is a study-specific measure (and is therefore relatively precise), the EAC is derived from an overall evaluation of a compound (i.e., using all available laboratory and field data). It is therefore subject to interpretation through expert judgement, and dependent on the risk scenario being evaluated (e.g., an EAC derived from a mesocosm study simulating a permanent Dutch ditch may differ from an EAC derived to protect ephemeral streams in the Mediterranean).

3.1.5 Implications of the FOCUS surface waters approach and considerations for landscape-level assessments

The implementation of the FOCUS water scenarios has a number of consequences for the development of higher-tier aquatic risk assessment. These are introduced below and then discussed in more detail in the following sections of the chapter. Approaches for considering a water body as a part of the landscape that includes other water bodies, and cropped and non cropped area will be considered, including:

- the hydrology of the water body and its influence on the ecology of the system
- the spatial distribution of water bodies and the hydrological network (influencing dilution and organism movement)
- the influence of temporal variation in chemical and organism dynamics.

It is envisaged that integrating landscape related factors into the risk assessment could improve it in a number of ways e.g.:

- by providing a better assessment of exposure, since aquatic ecosystems are no longer considered as isolated ones so that exchanges of material may occur
- by providing a better assessment of exposure within time, for similar reasons
- by defining the attributes of the ecosystem to protect, defined for specific water bodies
- by allowing a more comprehensive assessment of recovery potential.

3.1.5.1 Development of ecological characterisitics for ponds, ditches and streams

The Focus Surface Water group recently developed surface water exposure scenarios that include three types of water bodies (pond, ditch, stream) which are summarized in Table 3.3. These water bodies are relatively simple in that they are effectively two compartments i.e., a water phase of varying hydrology and a sediment phase. Although this development is an improvement from the previous approach of using a one-dimensional (30 cm depth) static water body, it raises a number of other considerations. Hydrological and morphological characteristics are known to have a profound influence on biological assemblages. Other local habitat factors may also be important. Consequently, it may be possible to use the characteristics of the water bodies defined by the Surface Water group to begin to parameterise the ecological characteristics of the scenarios. For example, by considering the climate, slope and soil type, it could be possible to develop a preliminary view of the sorts of species assemblages that may be associated with these water bodies. It is also apparent from the literature that a range of other factors influence assemblage composition. These water bodies also occur in different ecoregions (as defined under 2000/60/EC), so it would be anticipated that there will be differences in organism assemblages for biogeographical reasons.

Table 3.3: Association of water bodies with Step 3 scenarios, adapted from Doc Sanco/4802/2001

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Scenario	Weather station	Water body type(s)	Slope (%)	Soil type	Ecoregion*
D1	Lanna (SE)	Ditch, stream	0 – 2	Clay	Central plains
D2	Brimstone (UK)	Ditch, stream	0 – 2	Clay	Great Britain
D3	Vredepeel (NL)	Ditch	0	Sand	Central plains
D4	Skousbo (DK)	Pond, stream	0 – 2	Light loam	Central plains
D5	La Jalliere (FR)	Pond, stream	2 – 6	Medium loam	Western plains
D6	Thiva (GR)	Ditch	0 – 4	Heavy loam	Hellenic western Balkan
R1	Weiherbach (DE)	Pond, stream	2 – 4	Light silt	Central highlands
R2	Porto (PT)	Stream	10 – 30	Light loam	Iberic-Macaronesian region
R3	Bologna (IT)	Stream	0 – 155	Heavy loam	Italy, Corsica, Malta
R4	Roujan (FR)	Stream	2 – 10	Medium loam	Western highlands

* From WFD 2000/60 Annex XI

3.1.5.2 Relating the exposure profile to potential for effects

The exposure scenarios developed by the Focus Surface Water group include exposure profiles through time from the date of application and the weeks thereafter. These exposure profiles integrate different input events into surface water i.e., spray drift at the time of application and subsequent runoff or drainage events. When relevant, successive applications are integrated for exposure calculations. Such time-varying exposure has consequences for risk assessment. For example, refinement of the risk assessment beyond the use of the PECmax could then include considerations of the influence of concentration peaks and their subsequent dissipation, along with the duration or return frequency of those peaks in relation to appropriate ecotoxicological threshold concentrations.

Directive 91/414/EC requires that both short-and long-term risks to aquatic organisms are assessed. Acute (short-term) risk and long-term risks may be of very different nature due to the endpoints that are measured and the exposure conditions in the tests. Acute risk is often based on effects observed in short duration studies (from 1 to 5 days). Exposure conditions in the studies are maintained and the effect concentration derived from the study is compared to the PECmax. When the PECmax is derived from a narrow concentration peak, i.e. for substances that dissipates rapidly from the water column, the return frequency of the peak should also be taken into consideration when considering the potential duration of effects.

Assessing long-term risk is more problematic since test concentrations are constant over the relatively long duration of the study (from 21-28 to ca.100 days or more for laboratory studies). For compounds that dissipate rapidly or moderately, it is more difficult to assess the implications of chronic toxicity data. A variety of approaches are available to address these issues, and these are discussed further below.

3.1.5.3 Influence of landscape-related factors

At present, risk assessments for PPPs are conducted at the "edge-of-field" scale, meaning that risks are evaluated for a water body located at the edge of the field on which treatment(s) is (are) applied. The "edge-of-field" scale is often seen as a lower tier for the risk assessment (compared to the more realistic "landscape" scale which is considered as a higher tier) because it proposes a simplified pattern to evaluate risks to aquatic organisms, namely:

- input in the water body of actives substances and/or formulated products come from **one source** *i.e.* treatment of a single field area;

- **plant protection products are considered independently**, *i.e.* assessors have to answer to the question of risks posed by treatment of a crop with the product for which authorisation is asked for in the context of Directive 91414/EC (EU, 1991); they are not considered in the wider context of plant protection programs associated to crop management (e.g. tank mixtures or in a wider frame mixtures that may occur in water bodies due to multiple input sources);

- **the water body considered is hydrologically and biologically isolated**, *i.e.* no exchange or renewal of water and/or biological material is possible;

- **aquatic assemblages are considered theoretically** *i.e.* assemblages are presumed to be made up of sensitive species of primary producers, primary consumers (invertebrates) and predators (fish), irrespective of climatic or hydrological conditions.

These assumptions may be refined in order to assess the risks in a more realistic way, for example by:

- considering input from several sources (*i.e.* from several fields);
- considering the possibility of and assess risks in the context of multiple exposure;
- considering the degree of hydrological and biological connectivity of the water body (*i.e.* take into account that in a stream water is renewed, consider the possible arrival of aquatic organisms from upstream or from other and non hydrologically connected water bodies);
- considering an aquatic assemblage as more closely related to the water body type (e.g. consider the probability of macrophyte development, presence of fish).

It is important to note that it may be relevant to refine any of these aspects even at the "edge-of-field" scale. As discussed above, the ecological characteristics of the water bodies described in Step 3 could be considered. Furthermore, exposure to several active substances at the same time may occur even in isolated water bodies because of the use of "premix" products, but also when surface water is exposed to several substances over time. Moreover, in the case of ditches or streams, inputs may come from several treated areas located in the vicinity.

For aquatic species, connectivity of the ecosystem is an important consideration, even at the edge-of-field scale. Mobility of organisms allows recovery potential from an external source *i.e.* from another section of the water body and some species have mobile aerial stages so that even populations in an hydrologically isolated water body may recover due to external input of organisms.

When evaluating risks across a landscape or region, risks to the various surface water bodies may vary according to differences in exposure and differences in the organisms present. In this case, it may be appropriate to consider risks to populations at the meta-population level (i.e. by evaluating population dynamics across a range of water bodies and taking into account the potential movement of individuals between those water bodies). Furthermore, at a regional landscape level, it is also necessary to consider the influences of multiple exposure through time and multiple sources of exposure. These considerations regarding potential influences of scale on the risk assessment are summarised in Table 3.4 below. Performing risk assessment at the "edge-of-field" or at the "landscape" level should refer to the scale on which the risk is focused, rather than to the scale at which each parameter exerts its influence.

Table 3.4: Landscape factors that should be considered for the "edge-of-field" and at the "landscape" levels.

Landscape related factors	Scale of the risk assessment	
	Edge-of-field	Landscape
Ecological composition:		
- from water body hydrology	X	X
- from water body connectivity	-	X
Water quality restoration capacity:		
- from active substance information on fate and behaviour		
- from hydrological networks	X	X
	-	X
Ecological recovery:		
- from water body capacity to recover	X	X
- from ecological connectivity	(X)	X
Multiple exposure:		
- from the "edge-of-water body" area	X	X
- from all water bodies of the landscape	-	X

3.2 Factors that influence organism composition

In this section, an overview is provided of current knowledge concerning approaches for predicting the occurrence of species and communities in surface water, and their potential application in the development of ecological scenarios. The occurrence of species is principally determined by two factors: (i) the abiotic and biotic conditions which determine whether a species can inhabit the habitat concerned and (ii) the wider geographical distribution of species (biogeography). A review of these factors and their implications for risk assessment is described below. Suggestions for future developments are also included.

3.2.1 From conditioning factors to biological indices

3.2.1.1 Hierarchy of factors influencing freshwater assemblages

Organism assemblages are determined by a wide variety of factors operating at many scales. At the highest level, environmental factors like climate, parent substrate material and geomorphology will determine the occurrence of species. At a more local level, assemblages will be influenced by a variety of factors that are both spatially and temporally variable (see Table 3.5). The challenge for developing ecological scenarios would be to categorise those factors that are most relevant for distinguishing species assemblages that are likely to occur in agroecosystem landscapes.

A broad classification scheme used to derive species composition is described in the EU Water Framework Directive (EU, 2000), where, based on the work of Illies (1978), a number of European ecoregions for surface waters are defined (see Annex 3). Within these regions, the water bodies are characterised using altitude, size and geology. Altitude, size and geology influence the environment at the local scale via influences on soil structure, organic matter content, salinity, acidity, moisture, nutrient availability, substrate, saproby, dynamics, food source, micro climate, current velocity dimension and drying. These factors themselves then determine the species that are present. A ‘first cut’ at deriving ecological scenarios could therefore use such a classification (see also Table 3.3), whereby the relevant ecoregion for each of the fifteen water bodies in the ten surface water scenarios could be used as the starting point for defining the likely assemblage. Having established the broad characteristics of the scenario based on climate, substrate and geomorphology, it may then be possible to evaluate more local conditions described by the FOCUS surface water scenarios to estimate the species that would be expected to occur (e.g. by differentiating between species that typically occur on clay or sand substrates). Further information on how local factors might influence the assemblage is described below.

Table 3.5. Hierarchy of abiotic factors influencing water assemblages.

Major influence	Subtopic	Detail	Minutiae
Latitude, longitude, altitude	Climate	Rainfall, wind, isolation	Growing season; mixing cycles; seasonal cycles; permanence
Underlying geology	Water pH, sediment input	Links to topography	Transparency
Topography, slope	Water body morphology, catchment size	Depth, flow, basin contours, surface areas	Light penetration, oxygen distribution, littoral development
Human impact	Land use, drainage, sources of pollution	Impacts of addition of nutrients and toxicant	Temporal variation in concentration

3.2.1.2 Local assemblage conditioning factors and models

Predicting the occurrence of species based on ambient conditions is a fundamental aim of ecology. For aquatic organisms, much research has been conducted to determine the influence of abiotic factors on community structure. What emerges from such studies is that habitat type tends to determine the biological traits of organisms (and hence species) that live in them (the habitat templet or template theory of Southwood, 1977). A number of studies have demonstrated links between the species present and factors such as flow and substrate types (Statzner et al., 1997; Townsend et al., 1997). These have indicated that if the local habitat conditions are known, then the likely life-history attributes of organisms living there can be predicted, and with sufficient biogeographical information, likely species composition can be assigned to the water body.

There are many examples of studies where local habitat conditions have been used to develop assemblage prediction systems. A number of these are under development under the EU Water Framework Directive (EU, 2000), for instance the AQEM assessment system for riverine macroinvertebrates (see www.aqem.de), the StaR – Standardization of River Classification Project (see www.eu-star.at) In various Member States, prediction systems exist or are under development (for instance MOVE for aquatic vascular plants (*cf.* Bakkenes *et al., 2002)*, RISTORI for aquatic macrofauna in the Netherlands (Durand and Peeters, 2000, Verdonschot et al., 2003), RIVPACS for macroinvertebrates in the UK (Wright et al., 2000), PSYM for ponds in the UK (http://www.brookes.ac.uk/pondaction/PSYM2.htm), small riverine fish (Mastrorillo et al., 1997), plants and macroinvertebrates in ditches, streams, ponds and rivers in agricultural areas (Biggs et al., in prep), and the Illies classification of European limonfauna (Illies, 1978).

In some cases, these classification systems contain very detailed data on the range of physico-chemical conditions under which organisms are found to occur. One example of the type of data available is the Limnodata Neerlandica, a database consisting of aquatic organisms (phytoplankton, vascular plants, epiphytic diatoms, zooplankton and macro-invertebrates) and the ranges of abiotic parameters under which they occur (oxygen (mg/l), oxygen saturation (%), biological oxygen demand (mg/l), NH4-N (mg/l), Kjeldahl-N (mg/l), NO2-NO3-N, total N, ortho-P, total-P, pH, chloride, conductivity, Ca, Na, K, Mg, HCO3, SO4, chlorophyll, depth, area, time of year (month), soil type, water type, stream velocity, detritus layer). An example is given for *Daphnia pulex* (STOWA, 1997) in Table 3.6. Such data sets have clear applications in the development of ecological scenarios.

Table 3.6. Example of available data concerning the range of abiotic factors and the occurrence of species.

Daphnia pulex

Parameter	Unit	N	Average	P10	P90
pH	-	352	7.94	7.74	8.15
Conductivity	mS/m	48	64.1	33.6	114
View	m	41	1.35	0.25	3
Depth	m	303	2.26	1	3
Width	m	303	37.9	10	115
O_2	mg/l	50	9.31	3.65	14.3
O_2 saturation	%	9	93.3		
BOD	mg/l	31	5.79	1.9	9.3
P-tot	mg P/l	51	0.498	0.05	0.68
ortho-P	mg P/l	49	0.182	0.01	0.466
N-tot	mg N/l	22	1.55	0.715	2.69
Kjeldahl N	mg N/l	46	2.74	0.675	5.2
NH_4	mg N/l	83	0.365	0.035	0.68
NO_2+NO_3	mg N/l	161	1.36	0.11	2.94
Chlorophyl-a	µg/l	47	33.2	2	101
K	mg/l	21	7.18	4.81	8.05
Ca	mg/l	157	80.3	57.3	103
Mg	mg/l	155	9.17	5.76	13.4
Na	mg/l	21	23.7	17	26.5
Cl	mg/l	354	96.2	37.8	131
SO_4	mg/l	25	57	27	68
HCO_3	mg/l	223	208	173	288

These data provide the basis for predictive models such as those mentioned above. For deriving a predictive model from this kind of data a lot of ecological theory and testing of the model is needed. Although most of these models are developed with the aim to predict the effects of environmental changes, or in some cases recovery of community structure, they can be used to predict the community structure that might be expected under the given circumstances.

3.2.1.2 Biological Indices

Another approach that was reviewed was the potential application of biological indices for the development of ecological scenarios. To inform monitoring programmes, species data are gathered with the aim of building biological indices to summarize the ecological status of the water body. Such indices have been developed on the basis that the presence and abundance of species in an aquatic system is related to morphological, physical, and chemical descriptors of aquatic systems. As an example, invertebrate indicator taxa that are used in France for streams are listed in Table 3.7.

Table 3.7. Taxa used in the IBGN system and indicating values. Each of these taxa has been described through its ecological traits as related in Annex 5 (from Tachet *et al*., 2000).

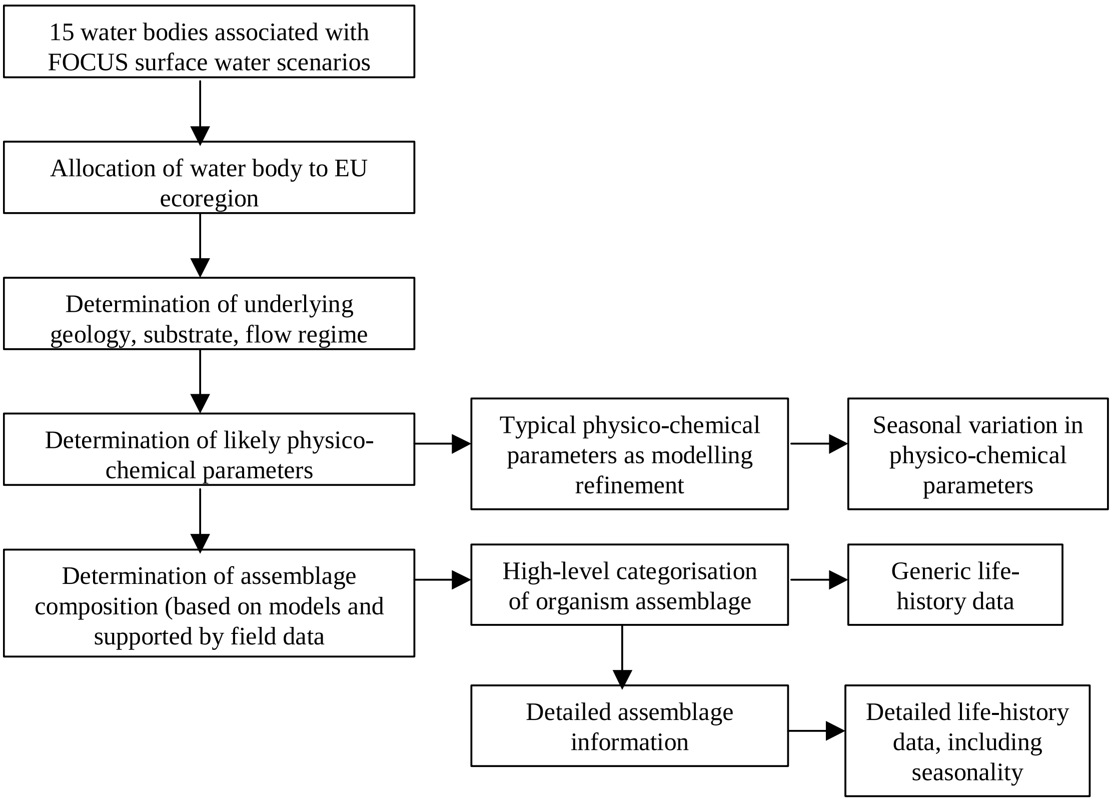
Indicating taxa	Value		Indicating taxa	Value
Chloroperlidae *Perlidae* *Perlodidae* *Taeniopterygidae*	9		*Leptoceridae* *Polycentropodidae* *Psychomyidae* *Rhyacophilidae*	4
Capniidae Braphycentridae *Odontoceridae* *Philopotamidae*	8		*Limnephilidae* *Hydropsychidae* *Ephemerellidae* *Aphelocheiridae*	3
Leuctridae *Glossosomatidae* *Beraeidae* *Goeridae* *Leptophlebiidae*	7		*Baetidae* *Caenidae* *Elmidae* *Gammaridae* *Mollusca*	2
Nemouridae *Lepidostomatidae* *Sericostomatidae* *Ephemeridae*	6		*Chironomidae* *Asellidae* *Acheta* *Oligochaeta*	1
Hydroptilidae *Heptagniidae* *Polymitarcidae* *Potamanthidae*	5			

Biological indices have been used in combination with chemical and physical indices to assess impacts of metal contaminants and physical alterations in urban streams (Rogers *et al.*, 2002), and of pesticides in running waters in agricultural areas (Liess et al., 2001b). Most often impact studies are based on the total number of taxa, the percentage of ephemeroptera, plecoptera and trichoptera (EPT) and the Hydropsychidae/Trichoptera ratio (Hickey and Golding, 2002; Rogers *et al*., 2002; Schmidt *et al*., 2002). Indeed, EPT are considered to present the highest diversity among invertebrates (Wasson, 1989). Moreover, the sensitivity of ephemeropterans and plecopterans to pesticides is often observed in microcosm and field studies (Schulz *et al*., 2002). Coleopterans also are considered relatively sensitive whereas *Dugesia* sp., chironomidae and simulidae are generally considered to be more tolerant to pesticide pollution. Here the remark has to be made that these conclusions are mainly based on tests with certain types of insecticides. For other groups of pesticides other sensitive groups might occur.

3.2.2 Future development of ecological scenarios and their use in risk assessment

Considering the available approaches mentioned above, it would probably be feasible in the future to begin to describe the communities of the different water body types included in the FOCUS surface water scenarios. A good example of a study which includes many of the approaches outlined below is that of the ‘Aquatic ecosystems in the UK agricultural landscape’ project funded by UK-DEFRA (Ponds Conservation Trust, 2003) which has described typical invertebrate and plant assemblages for a range of surface waters associated with UK agroecosystems. Such studies are recommended as a good point of reference for future work at the EU level. An overview of a potential approach to develop the scenarios is shown in Figure 3.2 and further details are described below.

Figure 3.2. Outline of a scheme for developing ecological scenarios for the FOCUS surface water scenarios



Of the many variables that influence the diversity of aquatic ecosystems, perhaps the key factors are biogeographical location, flow regime, and substrate type, and descriptions of these parameters are already available. With this sort of information, even if wholly empirical methods are not available, it is usually possible for the expert limnologist to predict the species that will be present. Based on the already available properties of the fifteen water body / scenario combinations in the surface water scenarios, it therefore seems likely that it would be possible to define ecological assemblages.

The level of detail of assemblage definition that would be achievable would vary among taxonomic groups and types of water body. For example, for macroinvertebrates and macrophytes, there are probably sufficient data for small streams and ditches to generate such ecological scenarios, but data are somewhat more limited for ponds. A preliminary summary table of likely data availability and feasibility of collecting data for the different water body types and certain taxonomic groups is shown in Table 3.8.

Data availability tends however to be patchy in the different member states and depending on water body types. In order to develop the ecological scenarios, it would be necessary to establish a group of expert limnologists from the various regions of Europe, and perhaps an extended network of European experts for consultation and checking (e.g., via a distributed network as proposed by the FreshwaterLife project www.freshwaterlife.org).

Table 3.8 Indication of data availability and feasibility of collection for different taxonomic groups in different water bodies

Taxonomic group	Pond		Ditch		Stream	
	Availability	Feasibility	Availability	Feasibility	Availability	Feasibility
Fish	Poor/ moderate	Low	Moderate	Moderate	High	High
Macro-invertebrates	Moderate	High	Moderate	High	High	High
Zooplankton	Poor	High	Moderate	High	n.a.	n.a.
Macrophytes	Moderate	High	Moderate	High	Moderate	High
Phytoplankton	Poor	Moderate	Moderate	Moderate	n.a.	n.a.

n.a. = not applicable

As a first step in developing ecological scenarios, it may be possible to propose quite a broad assemblage classification in terms of the most abundant species groups present at a high level of taxonomic resolution, e.g. Cyprinidae, Salmonidae, Gammaridae, Baetidae, etc. Starting at this level and then building in more complexity could also perhaps lead to the development of different tiers of ecological scenarios, depending on the degree of risk assessment refinement that was appropriate. For example, if it could be stated for a pond scenario that fish were likely to be represented by cyprinids rather than salmonids, the relevant ecotoxicological data could then be used to refine the risk assessment.

One further possible approach to support this development would be to use available field data of species composition in agricultural areas (see for example Williams et al., in press) or to collect those data in the future (although it is recognised that the feasibility of this is varied – see Table 3.8). Constraints of such a method include the extent to which data could be extrapolated and that sites would need to be selected carefully so that assemblages were broadly representative of uncontaminated conditions.

A further step in the development of ecological scenarios would be to link life-history information to the species present. For example, the presence of different life stages during the season could be an important consideration because of the potential for differences in sensitivity. Information on the life cycle characteristics of an organism would also be helpful in establishing recovery potential.

At present, the FOCUS surface water scenarios are parameterised in such a way that a minimum water depth is maintained. However, many surface water bodies vary substantially in their seasonal hydrology, even drying out for a proportion of the year. A review of these types of water bodies is included in Annex 4. Further work in this area should consider the importance of seasonally intermittent water bodies.

Developing an ecological component to the surface water scenarios could be used to refine further the higher-tier risk assessment. By identifying the taxa typically associated with the scenarios, it would be possible to refine the risk assessment by focusing on those species that are likely to be of concern. This could assist in the interpretation of existing data (e.g. by examining the sensitivity of those species present or interpreting micro/mesocosm studies), and could also guide the development of new approaches such as ecological modeling (e.g. by using information on the life-history of such organisms to both refine the effects assessment and to make some forecasts of likely recovery rates from any effects – see Section 3.5).

Again the work could start based on available scenarios described at step 3. These scenarios probably do not cover all specific crop situations and neither do they cover all ecological situations (e.g. citrus or olive crops, drying ditches, etc) and new scenarios that might be developed should also be considered for ecological purposes.

3.2.3 Influences of species sensitivity

One implication of developing ecological scenarios would be that there may be differences in sensitivity between assemblages depending on their composition. A brief review of the current use and applications of species sensitivity considerations is included below.

3.2.3.1 Species sensitivity ranking

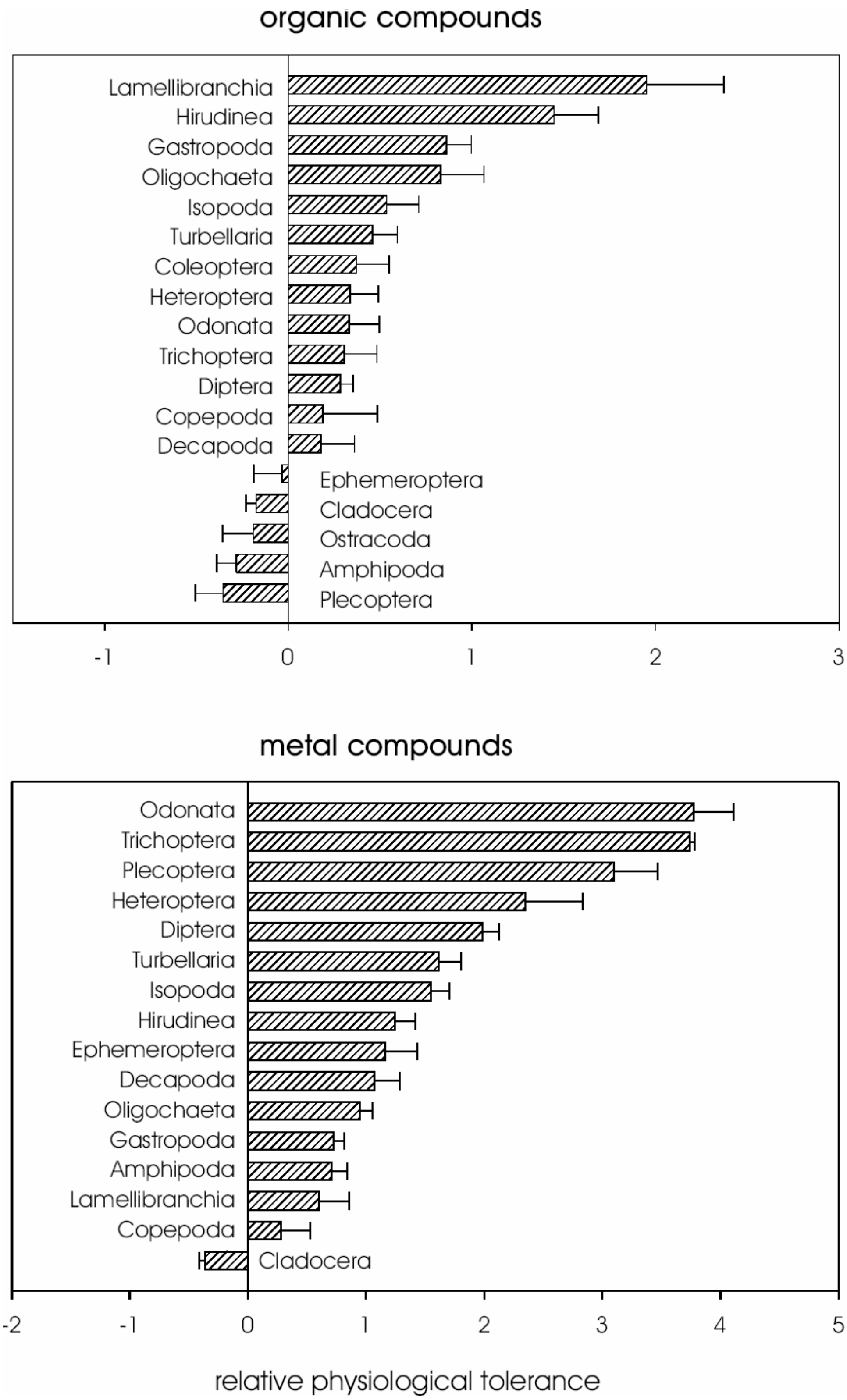
In the field, a range of species can be exposed to a variety of toxicants. To predict the effects of toxicants and to understand changes in species composition in communities it is therefore desirable to know relative species sensitivity to a range of toxicants. However, for many species little information about their sensitivity is available. Hence, a major problem of data limitation exists to predict the effects of toxicants (Posthuma et al., 2002). This problem in current risk assessment was also recognised during the HARAP workshop (Campbell et al., 1999).

The limitations associated with the use of standard test organisms lead to the application of species sensitivity distributions (SSD) in risk assessment. By the use of several test-organisms a distribution of sensitivity of organisms is developed to assess the fraction of species affected in the environment. This refinement of the concept of representative test species gives a better assessment of effects in the environment as the sensitivity distribution allows to estimate the affected percentage of the community. However, a major limitation of this approach is the extensive toxicological information required for the great variety of species assemblages existing in the field and the restriction to base the sensitivity distribution on one substance.

In a UK DEFRA project (DEFRA, 2002) it was indicated that within one taxonomic group, species from different geographical regions or freshwater habitats could be combined, meaning that influence of regions or habitat on the sensitivity is limited for the endpoints under consideration. Furthermore laboratory derived SSDs can be representative for (semi)natural assemblages. These conclusions are mainly based on experiments with insecticides and herbicides.

As one possibility to deal with the limitation of information and to include species data Wogram & Liess (2001) suggested to rank aquatic macroinvertebrate species according to their sensitivity using *Daphnia magna* as a point of reference as for this species a great number of toxicants are evaluated. This method enables the integration of various toxicants in the ranking of species as a relative sensitivity compared to *Daphnia magna* can be calculated for each species and toxicant. To ensure a sufficient statistical power, related species were aggregated and species were ranked separately with respect to organic compounds (Figure 3.3) and to metal compounds. This concept enables the ranking of a large number of aquatic macroinvertebrate species according to their sensitivity to these two groups of toxicants. The disadvantage of this concept lies in its reduced accuracy due to aggregation of information. Nevertheless, the authors successfully applied the method to predict the composition of the macroinvertebrate community in streams according to the pesticide contamination (Liess et al., 2001b). The dataset which formed the basis of this study was dominated by insecticides, so to some extent the conclusions are mainly limited to insecticides.

Figure 3.3. Differing relative physiological tolerances (log – relative sensitivity) of macroinvertebrate orders with respect to organic and metal compounds in comparison to *Daphnia magna.* The vertical line at x = 0 represents the tolerance of *D. magna*, with which the other values are compared. Details see Wogram and Liess, 2001.



Other species sensitivity ranking studies include those of Mayer and Ellersieck (1988), and Mohlenberg *et al* (2001). The main results of these studies together with the study from Wogram and Liess (2001), are summarised in Tables 3.9 and 3.10.

Table 3.9. Species sensitivity ranking among the invertebrate community (taxa appear from the most to the less sensitive).

Mayer and Ellersieck (1988)*	Mayer and Ellersieck (1988)**	Wogram and Liess (2001) organic compounds***	Mohlenberg et al. (2001)****
Claasenia sabulosa	Perlidae	Plecoptera	Trichoptera, Cladocera
Pteronarcys californica	Pteronarcidae	Amphipoda	Plecoptera
Pteronarcella badia	Gammaridae	Ostracoda	Hemiptera
Palaemonetes kadiakensis	Daphnidae	Cladocera	Ephemeroptera
D. magna, D. pulex		Ephemeroptera	Coleoptera
Gammarus fasciatus	Asellidae	Decapoda	Amphipoda
Simocephalus serulatus		Copepoda	Isopoda
Asellus brevicaudus		Diptera	Odonata
		Trichoptera	Gastropoda
		Odonata	
		Heteroptera	
		Coleoptera	
		Tricladida	
		Isopoda	
		Oligochaeta	
		Gastropoda	
		Hirudinea	
		Lamellibranchia	

*from 14 to 25 species being tested with 14 insecticides.
**idem, among families
***from arithmetic means of 5 to 460 individual values
**** insecticides

Mayer and Ellersieck also proposed the following table for fish (Table 3.10):

Table 3.10. Several species sensitivity ranking among the fish community (taxa appear from the most to the less sensitive).

Mayer and Ellersieck (1988)*	Mayer and Ellersieck (1988)**
O. mykiss	
L. macrochirus	
Micropterus salmoides	
O. kisutch	
Salmo clarki	Centrarchidae
Perca flavescens	Palaemonidae
Salmo trutta	Salmonidae
Ictalurus pyunctatus	Astacidae
Cyprinus carpio	Cypridae
Lepomis cyanellus	
Pimephales promelas	Ictaluridae
Cazrassius auratus	Cyprinidae
Ictalurus melas	

*from 14 to 25 species being tested with 14 insecticides.
** idem, among families

Among invertebrates, Wogram and Liess (2001) found that Cladocera are often among the most sensitive species to both organic and inorganic compounds and this is validated by other authors (van der Geest, 2000). The ranking of Mohlenberg *et al.* (2001) also indicates that Cladocera are among the the most sensitive species to some insecticides. Water fleas were also found to be the most sensitive species to some aromatic amines in SSD studies (Ramos *et al.*, 2002). However, the comparison is probably influenced by the fact that data on other invertebrates are still far less numerous than for Cladocera (Wogram and Liess, 2001). This might in part explain some variability between studies, e.g. the fact that Trichoptera were found to be as sensitive as cladocera in the study of Mohlenberg *et al.* (2001) but not in the study of Wogram and Liess (2001). One has also to consider that the differences in sensitivity among life-stages of these species, in combination with the fact that every taxa was not tested at the most sensitive development stage might have led to some bias in species sensitivity ranking (Stuijfzand, 1999).

According to Mayer and Ellersieck (1988), species sensitivity distribution of toxicity values may present various orders of magnitude depending on the class of the animals. The distributions of toxicity values (EC/LC_{50} for 352 chemicals, 61 species) were eight order of magnitude for crustaceans, seven for insecta, nine for osteithyes and four for amphibia.

Among species, frequency distribution ranged over nine orders of magnitude and 90 percent of the values fell within five orders of magnitude.

3.2.3.2 Relationship between biological indices and species sensitivity ranking

The polluosensitivity index based on invertebrate relative abundance data (described above) was compared to available sensitivity ranking to toxicants in order to evaluate the comparability of the two ranking methods Comparison of the IBGN polluosensitivity index and the database ranking of Wogram and Liess (2001) provided a poor correlation (see Annex 6). This was not surprising since the IBGN classification is based on non-specific compounds while the ranking of Wogram and Liess is based on sensitivity to active compounds. Furthermore, the level of identification was not similar in the two databases thus limiting analytical sensitivities. Finally, many factors probably influence the survival of a population in natural aquatic ecosystems which are not included in laboratory tests for species sensitivity ranking databases.

From the above data, it appears that species or taxa that are described as "indicator species" for organic pollution might not be considered *a priori* as relevant indicator species concerning pesticide pollution. This may be explained in part by the specificity of the modes of action of pesticides compared to general organic contaminants. Other factors such as individual/population mobility and behaviour in the aquatic system, as well as differences in sensitivity between different life-stages for a species have also been proposed (Stuijfzand, 1999). Therefore, predictions of communities as deduced from models based on abiotic factors may not rule out sensitivity of communities to active substances.

3.2.4 Conclusions

There is currently no ready-to-use model for developing ecological parameters to accompany the FOCUS surface water scenarios. There is however a growing body of data and approaches that could be used to begin to develop ecological surface water scenarios based on the properties of the ambient environment. The available models require at least climatic (e.g. temperature), hydrological (e.g. water regime) and substrate descriptors. Since highly detailed assemblage descriptions are currently less feasible, one preliminary option would be to develop a system at the ecoregion level at a low level of taxonomic resolution (e.g. family level).

Ecological scenarios could be used to help design and interpret higher-tier studies and risk assessments. For example, an ecological scenario could be used to evaluate recovery potential of a sensitive group under differing conditions. However, their development should certainly not be seen as a requirement for testing every assemblage type.

3.3 Factors that influence effects

As developed in chapter 3.2, species assemblages in aquatic ecosystems are the result of conditioning factors that are gathered under (i) climate, (ii) hydrology and (iii) nature of substrate, and commonly stated as temperature, pH, oxygen concentration, etc. These factors are discussed here as abiotic factors. Ecologisation of scenarios means to integrate the most important of them among descriptors of water bodies. The inference of that is then to identify whether abiotic factors may exert an influence on the overall observed effect of substances on aquatic organisms and if yes, to try to define methods to integrate this into risk assessment. In section 3.3.1, typical abiotic factors related to aquatic systems have been reviewed for their influence on the toxicity of chemical substances.

A special case of abiotic factors related to agricultural practices is the presence of other toxic substances, especially pesticides. They are discussed in section 3.3.2.

Other factors influence general impact of substances, which relate to organisms themselves. These are discussed here as biotic factors (section 3.3.3). Biotic factors are typically related to trophic and demographic aspects in communities. This review has been performed with the aim to get an idea of the relative importance of biotic factors in the global response of organisms to the presence of toxicants.

When abiotic factors also act indirectly on the response of an organism to the presence of a toxicant, through their influence on biotic factors, this is discussed in the chapter of the abiotic factor.

3.3.1 Abiotic factors

In this section, an overview is given for the abiotic parameters temperature, pH, O_2, organic matter, salinity, nutrients, drying and substrate for which data are found in the literature on their influence on the toxicity of pesticides. The actual values of these parameters in practice can be used to describe the potential effects of pesticides in an actual situation. This could result in “mitigation”: for instance, under certain circumstances the toxicity could be lower than predicted on the basis of lower tier data. However, the opposite could be the case as well. Below the abiotic parameters are discussed, including examples from the literature.

3.3.1.1 Temperature

According to the review of Mayer and Ellersieck (1988), temperature and toxicity are positively correlated for most chemicals. Reasons for temperature-dependant toxicity may be differences in respiratory rate, chemical absorption, and metabolism of chemicals, so both sensitivity of the organism and availability of the pesticide can be affected. Analysis of 48 chemicals and 90 tests at different temperatures with fishes showed slopes different from zero in 26 tests with 19 chemicals, the ratio between the highest and the lowest averaged 9.8 [3.1-51]. Reports addressing temperature differences showed that most chemicals, including inorganics, were increased or decreased within a two to four-fold factor in LC_{50} per 10°C change. Some explanation on the influence of temperature on toxicity of chemical compounds was provided by Del Vento and Jachs (2002) with models for bioaccumulation of POPs by bacteria and phytoplankton. Indeed, both uptake, adsorption, desorption and depuration constants entering the dynamic of bioaccumulation are temperature-dependant.

3.3.1.2 pH

The toxicity of chemicals (in particular weak acids and bases) may vary considerably with the acidity of the media. The reason is that unionised forms diffuse more readily in organisms. Analysis on 49 chemicals and 100 acute tests (pH range 6.0-9.5) showed slopes significantly different from zero in 23 tests with 10 chemicals, with an average ratio (lowest:highest LC50 of 16 [4.2-45] (Mayer and Ellersieck, 1988)). Slopes were consistent among species within a chemical, thus indicating chemical rather than biological differences. As an example, effects of triphenyltin and tributyltin on ATP synthesis of the purple bacterium *Rhodobacter sphaeroides* were conditioned by pH, inhibition being higher at low (6.1) than high (7.5) pH in the laboratory (Hunziker *et al.*, 2002).

3.3.1.3 Oxygen concentration

Although the oxygen concentration can affect the substance itself, the most important effect will by on the sensitivity of species. Oxygen concentration in water may be highly variable in rivers (van der Geest, 2000), from 10 to 50% and may vary within time. In his study, van der Geest 2000 showed that a lower percentage in oxygen saturation could, beside direct effects, influence the sensitivity of aquatic insect larvae to inorganic and organic chemicals. This was also observed by Stuijfzand (1999) with Hydropsyche in natural (river water) conditions. But inverse results may also be observed if for example, the chemical reacts with oxygen and generates more toxic daughter products (McClosey and Oris, 1991 in Van der Geest, 2000). Low concentrations of oxygen also increase the sensitivity of the cladoceran *Daphnia pulex* towards the pesticide carbaryl (Hazanato and Dodson, 1995) and towards organic chemicals in a model population of *Daphnia magna* (Koh et al., 1997). Finally, oxygen concentration may also influence the ability of a population to recover after a chemical stress. Indeed, several authors demonstrated that increasing oxygen levels in rivers could allow re-colonisation of the caddisfly *Hydropsyche contubernalis* (Neumann, 1994 and Becker, 1987, in van der Geest, 2000).

3.3.1.4 Inorganic nutrients

For all organisms, the trophic status also influences the sedimentation of abiotic and biotic particles and thereby the deposition of xenobiotic compounds due to sorption of organic and inorganic toxicants (Koelmans *et al.*, 2001 in De Haas *et al.*, 2002). A ratio of N/P below one was found to increase inhibitory effects of pendimethalin on the growth of the chlorophyceae *Protosiphon botryoides* (Shabana *et al.*, 2001). Low nutrient treatments were less favourable for recovery in natural periphyton exposed to herbicide stress, compared to higher nutrient conditions (Lozano and Pratt, 1993, in De Lorenzo *et al.*, 2001). Van den Brink *et al,* (2001) show combined effects of nutrients and chlorpyrifos. In this case however, these are *indirect* effects. The nutrients did not influence the toxicity of the insecticide, but the differences in effect in low or high nutrient environments were caused by different growth rates of primarily the algae.

3.3.1.5 Organic matter concentration

Studies on the toxicity of some inorganic pesticides and surfactants to *Echirogammarus tibaldii* showed that the presence of organic matter could reduce observed effects (Pantani *et al.*, 1995). Algae growth rates may also be slightly influenced by organic compounds in concentrated river water (Van Dijk *et al.*, 1995 in Stuijfzand, 1999). Suspended particles may influence the toxicity of organic pesticides, as shown on larvae of the trichoptera *Linephilus lunatus* exposed to water-dissolved or particle-associated fenvalerate in laboratory conditions (Schulz and Liess, 2001a). Differences in terms of exposure concentration amounted to a factor of 10 for the endpoints of temporal emergence and adult weight, and 100 for total emergence success and biomass production. Lethal and sublethal effects were expected to result from transient aqueous phase contamination while sublethal effects were expected to be related to short-term exposure to sediment–associated pesticide. Nevertheless, field relevant inputs of this compound were found to induce lethal effects (at 10 µg/kg bulk) and sublethal effects (at 1 µg/kg) in macroinvertebrate species (Schulz and Liess, 2001b). Another study in which isoproturon was seen to bioaccumulate in some plants showed that even bioaccumulated isoproturon in macrophytes could be considered to be able to constitute long-term exposure to non-target organisms in aquatic microcosms (Merlin *et al.*, 2002).

3.3.1.6 Drying

Some species are adapted to drying of streams in summer and emergence therefore occurs mainly in early summer, as is the case for the trichopteran *Limnephilus lunatus* (Liess, 1998). The author showed that sublethal effects of insecticides in this species, such as a delayed emergence in the year, could in fact result in lethal effects due to subsequent stream drying. Drought increases the sensitivity of spiders towards the pesticide deltamethrin (Everts et al., 1991). Species inhabiting temporal streams are characterised by a short generation time and therefore a high potential of recovery. An investigation of such streams revealed that even at relatively high concentration of pesticides no effects could be detected (Liess et al., 2001b). Lowered oxygen concentrations, changes in salinity and temperature can interact in a variety of ways and influence the response of aquatic organisms to pollution (van der Geest, 2000). In a recent review , DeLorenzo *et al*. (2001) discussed the influence of water quality on the impacts of pesticides to microorganisms. The authors cited pH, salinity conditions, nutrient conditions but also microbial density as potential parameters conditioning pesticide toxicity in water.

3.3.1.7 Abiotic factors in the risk assessment

Abiotic factors have been seen to play a role in the overall response of organisms to toxicants either by conditioning the fate of a substance (pH, oxygen concentration, organic and inorganic matter), or ADME[17] aspects (temperature, oxygen concentration) or population dynamic (temperature, oxygen concentration, organic and inorganic matter as nutrients) and even re-colonisation (oxygen concentration). In this way they may both act on the chemical and the organism, thus meaning that attempts to model the effect of a chemical in systems defined by one or several of these factors might become very complex.

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[17] Absorption, Distribution, Metabolism, Excretion

It is important to note that even when trends were observed (e.g. general increase of toxicity with temperature), it is in general difficult to generalise as relationships in many cases are substance-related, especially for fate, bioavailability and ADME aspects. Another point is that abiotic factors condition assemblages (Section 3.2) but also population dynamics (temperature, oxygen concentration, nutrients) in that way that they condition the evolution of assemblages within time. This might be an important aspect in developing models for effects of substances on populations that would take such factors into account.

As far as modeling is concerned, models have been developed to account for important abiotic and/or biotic factors when predicting growth rate and impact of chemical stressors on growth rate for aquatic invertebrates. The model of Waito *et al.* (2002) was in part developed with the aim to account for fate aspects in predicting effects. The authors proposed a model that uses typical laboratory data for linear alkyl benzene sulfonate (LAS) and parameters such as initial biomass, optimal consumption temperature, maximum consumption rate, specific dynamic action, maximum respiration temperature, respiration rate, excretion rate and mortality rate for species of zooplankton, benthic insects and invertebrates, omnivorous and piscivorous fish. The model derives benchmark levels that are approximately one order of magnitude less than the field derived NOEC found in the literature for LAS.

In a recent paper, Ringwood and Keepler (2002) proposed growth rate plots for sediment organisms (juvenile clams *Mercenaria mercenaria*) as a function of average overlying dissolved oxygen concentration, salinity or pH. The level of effect of water contamination on growth could then be estimated by comparison of measured populations with model estimates. Such approach may be useful to distinguish between environmentally induced decrease in growth and chemically induced decrease in growth.

Extrapolation of a particular assessment to the whole community level from standard laboratory tests is more complex. The diversity of interactions implied has led authors to develop community models (Norton *et al.*, 2002; Waito *et al*., 2002). The possible use of this type of model was not assessed in the current review.

As far as risk assessment is concerned, it is recognised that fate of chemicals and responses of non-target organisms in risk assessment need to be integrated (e.g. Belanger *et al*., 2002, Cuppen *et al*., 2002, Van den Brink *et al*., 2002). Among the reasons for this is the relationship between the outcome of studies and their design, the relative importance given to autotrophic organisms and invertebrates, and the bioavailability of chemicals and exposure profile may have consequence of the final effect chain (Belanger *et al*., 2002). It has been seen that abiotic factors such as pH, oxygen concentration, temperature, may influence the response of aquatic organisms to chemical stressors by conditioning the bioavailability of chemicals to organisms. In general this aspect of toxicity is considered to be covered in laboratory tests since media conditions are gathered to maximize exposure to the substance.

Abiotic factors may also act directly, e.g. on species metabolism or reproduction rate, hence conditioning the overall response of organisms. Again laboratory tests tend to maximize this aspect since conditions are gathered in order to optimize metabolism and reproduction rate. Abiotic factors finally define abundance and diversity of flora and fauna that may be encountered. In this context, it seems important to account for typical physical and chemical values for each of the three receiving water bodies. As an example, typical values for pond, ditch and stream could be compared with those recommended in standard toxicity tests in attempts to refine first tier risk assessment.

3.3.2 Other active substances

The EU system of pesticide registration is aimed at the registration of a single active ingredient and a representative formulation. This means that the potential effects of the use of other pesticides in the same area and time are not assessed. However, risk assessment for aquatic organisms exposed to pesticides at the landscape-level (see Section 3.1) implies that the assessor is able to give a clear idea of the impact of agricultural practices at a local scale, taking into account expected exposure of local organisms to active substances and eventually metabolites that were found to be relevant. Although this type of question is not incorporated in the registration process, in this chapter the subject is discussed briefly. Since at the moment the legal grounds are lacking, no proposal for methods to incorporate mixture toxicology in the registration procedure is given.

Mixtures of pesticides refer, at the landscape level, to various types of practices and situations, which may be grouped as below:

- Tank mix: two or more different pesticides are applied at the same time.
- Crop mix: in the same crop several pesticides are used in a certain period, resulting in a mix of pesticides in the surface water.
- Area mix: in a certain area in all crops different pesticides are used, resulting in a complex mixture of pesticides in different concentrations in the surface water.

- Surface water mix: a combination of the area mix with import from other areas, for instance from upstream sources and atmospheric deposition.

3.3.2.1 Tank mix

Tank mixtures present a particular status from the regulatory point of view. For the registration at present, attention is only paid to mixtures when a mixture is applied as such i.e. a formulation pre-mix (see for instance Godson et al., 1999). The current registration process for PPPs is based on active substances and their related representative preparations (Directive 91/414/EC, (EU 1991)). The latter may contain mixtures of one or 2/3 active(s) substance(s) with one or several adjuvants. For the registration of mixtures of active substances, data for mixtures are needed if it is not possible to predict its toxicity from the individual active substance data. It is stated that this is especially the case if a product contains two or more active substances (Section 10 of Annex III of the Authorisations Directive (Directive 96/12/EC, (EU, 1996)). Implicitly this means that it may not be possible to predict the effects of a mixture of active substances and the use of mixtures is in principle not "authorised" when the risks are not evaluated. A special case is the active substance manufactured as a mixture of isomers. Magrans *et al.* (2002) propose a procedure for the assessment of these type of active substances. Nevertheless, tank mixtures are in some cases included in practical recommendations, mainly with the aim to save time, limit the number of passages of tractors, or for efficacy purposes. Some widely recognised recommendations with regard to tank mixtures are summarised in Table 3.11.

Table 3.11. Examples of recommendations about the practice of tank mixtures (adapted from Monnet, 2002).

Criteria	Recommendation
Number of active substances	Mixtures containing more than three products should be avoided
Type of active substances	Mixtures of more than two insecticides or two fungicides or two herbicides should be avoided
Application rate	Tank mixtures of two herbicides should lead to reductions of application rates
Specific activities	Mixtures with acaricides should be avoided for efficacy purposes
	mixtures of two products of similar mode of action should be avoided

Tank mixtures are for example currently practised in vegetable crop protection, such as mixtures of two fungicides or two insecticides of complementary action; these products may also be applied at the same time (e.g. spinach or lettuce). Herbicides also are applied mixed together or with fungicides and/or insecticides. In cereals, the most frequent tank mixtures are made with 2/3 fungicides, 2/3 herbicides, herbicide + insecticide (2 or 3 actives substances), herbicide + fungicide (2/3 actives substances). However, mixtures may contain more (5/6) active substances in some crops (sugar beet for example).

3.3.2.2 Methods for assessment of the effects of mixtures

A number of methods are available for assessing the effect of mixtures:

Concentration addition

Concentration addition usually is used to describe the toxicity of a mixture of compounds with the same mode of action. The model can be described with the equation:

$$C_{eff} = \sum(C_i/EC_i)$$

In which:

C_{eff} is the overall effective concentration of a mixture

C_i is the actual concentration of the individual compounds (i)

And EC_i is the effect concentration of an individual compound, for instance EC_{50} for Daphnia.

The quotient C_i/EC_i is also referred as the "Toxic Unit" and C_{eff} as the sum of the Toxic Units.

Since additivity of Toxic Units assumes that the substances have the same mechanisms of action, the proportions of the toxic units can be added arithmetically. Comprehensive examples for the use of Toxic Units in evaluating risk posed by core active substances to aquatic organisms in rivers are provided by Steen et al. (1999) and Battaglin and Fairchild (2002).

Effect or response addition

For independent action the following equation is proposed (Vighi et al., 2003):

$$f_{(1,2,\ldots n)}=1-[(1-f_1)(1-f_2)\ldots(1-f_n)]$$

In which f_i are the fractions of the total possible effects of the individual toxicant i.

Synergy and antagonism

As summarised by Könemann and Pieters (1996), synergistic effects indicate either kinetic or dynamic interactions of implied components.

For all three types of models examples from the literature are available. For concentration addition a number of examples with compounds with comparable modes of action exist. Bailey et al. 1997 found additivity for insecticides chlorpyrifos and diazinon, tested with Ceriodaphnia dubia. In a microcosm study the combined effects of lindane and chlorpyrifos were studied (Cuppen et al., 2002; van den Brink et al., 2002). The results of this study indicated that at the community level the combined effects could be predicted on the basis of the toxicity of the individual compounds; the overall risk assessment indicates no antagonistic or synergistic effects of the mixture at the ecosystem level. A comprehensive review of existing studies is provided in ECETOC (2001). In general, conclusions meet those of Deneer, as regards the additivity of mixtures. Deneer (2000) assessed the usefulness of the concept of concentration addition for the joint effect of pesticides on aquatic organisms, using literature data from 1972 to 1998. The study showed that in 90% of the 202 mixtures - of two PPP or more - studied, concentration addition appears to predict the effect concentrations within a factor 2. In the dataset fungicides were poorly represented and data were dominated by effects of insecticides for fish, crustaceans and insects, and by herbicides for algae. The results were found for compounds with similar modes of action as well as for compounds with dissimilar modes of action. Hermens et al. (1984) and Hermens and Leeuwangh (1982) reported that the toxicity of mixtures of substances “with various structures and probable modes of action” was predictable with the concept of concentration addition. The analysis of combined effects among 137 binary mixtures of 14 pesticides and 5 surfactants, using algal biotests, showed that response addition as a reference leads to an underestimation of expected mixture toxicity for almost all combinations, irrespective of the type of mixture, while concentration addition provides a reference where over- and underestimates of mixture toxicity are balanced (Altenburger et al., 1996). Observed mixture toxicity deviated only in rare cases by more than a factor of 2 from predictions (i.e. within experimental variance).

Other studies are more in favour of the effect addition model. Faust et al. (2003) tested the joint algal toxicity of 16 dissimilarly acting chemicals. The results of this study show that concentration addition overestimates the toxicity with a maximum factor 3.2. Walter et al. (2003) investigated the toxicity to the algae Scenedesmus vacuolatus of a mixture of 11 structurally dissimilar substances at their NOEC concentrations. Results showed that a combined effect is observable which shows a higher intensity than that of any individual compound. The magnitude of this effect was more precisely predicted by the model of independent action than by concentration addition.

For synergistic effects examples are available as well. Some cases of synergism are well described, such as fungicides inhibiting sterol biosynthesis. Substances of this type inhibit the lanosterol-14α-demethylase, involved in the synthesis of sterols building the cell wall of the fungi. The mechanism relies on an imidazole group in the molecule. This imidazole has a high affinity for cytochrome-P450, which leads to inhibition of these ubiquitous enzymes also involved in the metabolisms of xenobiotics. This synergism occurs only when the exposure is simultaneous. Synergistic effects of atrazine on organophosphate toxicity to terrestrial and aquatic invertebrates have been reported (Anderson and Lydy, 2002; Jin-Clark et al., 2002).

Synergy is not always related to similarity of target but may be related to modified uptake as shown in a mesocosm study where Lytle & Lytle (2002) found an influence of a mixture of atrazine plus chlorpyrifos on the uptake of chlorpyrifos by the freshwater macrophyte Juncus effusus, but no effect on the uptake of atrazine was noticed. Cleuvers (2003) also described increased toxicity of mixtures of pharmaceuticals to Daphnia despite theoretically different modes of action.

3.3.2.3 Indications for the effects of mixtures

The method of concentration addition was applied to pesticide use in the 15 most important crops in the Netherlands (Deneer, 2003). Only pesticide drift, calculated according to realistic worst case" assumptions, was taken into account as an exposure route. The results of this study show that for 13 of the 15 crops the value 0.1 Toxic Units is exceeded. This is the level at which ecosystem effects were presumed to occur. Sixty percent of the exceedance of the 0.1 level is caused by a combination of different compounds used within the same crop. In Switzerland Chèvre (2004) presented a tiered approach, distinguishing smaller and larger water types. The combined exposure to agricultural and non-agricultural herbicides indicates a risk for over a three-month period in the larger water bodies. In two semi-field experiments a crop the application regime for potatoes and tulips was simulated. At exposure levels to be expected in the field, no indications for mixture effects were found (Brock, 2003).

SSD's were used to predict threshold values for toxic effects in sediments contaminated with mixtures (Fuchsman *et al.*; 1999). LC_{50}, Kow and acute to chronic ratio where available were plotted as SSD. The model derived "predicted effect threshold" (PET as an exponential function of Kow divided by the ratio LC_{50}/ACR). From this model, authors defined 20th

percentile PET that were one to 2 orders of magnitude higher than published screening level guidelines for chlorinated benzenes, and were consistent with relevant spiked sediment toxicity data.

3.3.2.4 Multi-exposure and risk assessment

As stated in the introduction of this chapter the system of pesticide registration is aimed at the registration per compound. As a consequence, the potential effects of the use of other pesticides in the same area and the same moment are not assessed. Therefore, the subject is mainly out of the scope of the present FOCUS group, and no recommendations for incorporating mixture toxicology into the registration procedure are made.

As shown in this section, models to predict the toxicity of mixtures of xenobiotic have been developed, mainly with the aim to predict the toxicity of mixtures of active substances. In the context of this document, these models should apply in order to solve problems related to mixtures of active substances and their metabolites, which are raised when toxicological data for the active substance and/or its metabolites are deduced from experimental conditions that do not allow to cover multi-exposure. These models may also apply when registration for a product with different active substances is applied for.

Concerning the different models, the most practical approach seems to start with the assumption of the concept of concentration addition. For this concept only EC_{50} (or EC_{10} or NOEC) data are needed and the concentration - effect quotients for the different compounds are added. It is noteworthy that the models apply for data that have been generated under similar experimental (exposure duration, exposure regime) conditions and for the same species. With the available information from exposure modelling (see other parts of this report) it is possible to have data concerning the different concentrations at a certain moment. To assess the effects of mixtures at a higher level of ecological functioning the models as described are not appropriate. This kind of effects are only taken into account in higher tier studies, containing trophic networks, as is the case in mesocosm.

From the review of mixture toxicology it can be concluded that, although discussions about methods are still going on, the available methodology would allow the inclusion of mixture toxicology into risk assessment. Based on literature and theoretical approaches, the presence of a mixture of compounds in the field can lead to higher risks. Considering uses at the scale of a field, it is unlikely however that similar concentrations of substances being of high risk and presenting similar properties (thus being susceptible to clearly increase the overall risk compared to that from a single compound) occur at the same time at a location. Field studies aimed at mixture toxicology are scarce, and yield little evidence for the occurrence of these effects. Therefore it is recommended to investigate the actual occurrence of mixtures in practice, together with the actual effects of these mixtures. In case such a survey indicates a potential risk, the incorporation of these aspects in the registration procedure should be considered.

3.3.3 Biotic factors

In this section, an overview is given for those biotic parameters for which literature data are found of their influence on the toxicity of pesticides. Table 3.12 summarises the results:

Table 3.12. Summary of biotic parameters influencing the toxicity of pesticides.

	Important	Data available	Remarks
Food	Y	Limited	-
Predation	Y	Limited	Increases effect of insecticides
Competition	Y	Limited	-
Sensitivity of species related to frequency and exposure duration	Y	Y	Different reactions depending on substance and species
Life stages	Y	Y	Generally young life stages more sensitive

3.3.3.1 Food

As far as invertebrates are concerned, exposure to aquatic toxicants may induce negative effects in sensitive species due to intrinsic toxicity, but it may also result in abundance peaks within the most tolerant species due to abundance of food, because of a loss of competitors. Effects on survival and growth in *E. virgo* appeared to be deeply related to the presence of toxicants in the sediment, and increased food concentration in general did not influence toxicant impacts. In contrast, *C. riparius* survival and growth responded to quality of the sediment rather than to toxicants. It was concluded that the trophic state of an ecosystem could influence the ecological risk of toxicants in a species-specific way. Assimilation of both metal and organic contaminants in aquatic invertebrates is a function of food quantity and quality (see Wang and Fisher, 1999, for a review). Wang and Fisher (1999) reviewed the relative importance of metal uptake from dissolved and food phase for aquatic invertebrates.

3.3.3.2 Predation

Predation has also been seen to increase the adverse effects of some insecticides to arthropod communities. The effects of the presence of fish (*Galaxias zebratus*) on *Baetis* populations exposed or not to azinphos methyl or fenvalerate at a sublethal concentration were studied in two artificial streams (Schulz and Dabrowski, 2001). From their results, it appeared that the combination of fish presence and insecticide exposure led to a significant increase in the mortality rate (9% when exposed to azinphos methyl and 25% when exposed to fenvalerate) compared to when a 0.8% mortality in mayflies exposed to either one or the other stressor. Mortality was then from 10 to 30 fold higher than in the control-fish treatment (Schulz and Dabrowski, 2001).

However, numerous ecological studies on predator prey relationships show regulatory capacities of such interactions in the long term (Begon et al, 1990). Examples include increasing growth rate of preyed populations as density controlled regulation is decreased and the shift of the predator to other prey when the prey population is strongly reduced.

3.3.3.3 Competition

Competition within and between species is a very important factor influencing the sensitivity towards toxicants. Due to compensation processes a population with high competition for food or space shows a reduced survival only at concentrations of 3 orders of magnitude higher than a population with low competition, as it was observed for trichopteran *Limnephilus lunatus* exposed over a short term to fenvalerate in laboratory conditions (Liess, 2002).

3.3.3.4 Sensitivity of species related to exposure frequency and exposure duration

Partial Least Square (PLS) regressions based on characteristic from 112 mesocosm experiments revealed that in general, at a given total dose the effect of pesticides decreases with number of pesticide additions. This means that in these cases a low but persistent pesticide concentration will have a lower effect on the macroinvertebrates than a high but temporary pesticide concentration (Mohlenberg *et al.*, 2001). The mode of action will be very important for determining effects, however. As for zooplankton, frequent dosing prevented algae recovery. Kallander *et al.* (1997) showed that pulsed exposure to insecticides was less toxic to the midge *C. riparius* if recovery occurred between the exposure events. The length of recovery period depended on the mode of action of the insecticide.

Several studies with algae show that the frequency of contamination may induce tolerance in exposed populations, as observed for algal population of outdoor mesocosms exposed to atrazine (Seguin *et al.*, 2002). Short-term sensitivity to atrazine in control (unexposed) communities was correlated with biomass estimation, diversity, oxygen concentration, nutrient concentrations and water temperature, whereas short-term sensitivity in formerly exposed communities was correlated with the concentration of some nutrients and water temperature. This meant that if environmental factors and community have a significant impact on tolerance induction, the impact of the chemical on tolerance induction was greater (Seguin *et al*., 2002).

Stuijfzand (1999) showed that toxicity of pesticides could increase with exposure time for first and fifth larval instars of *C. riparius* and *H. angustipennis*, which has been observed for other pesticides and other non-target species (Hutchinson *et al.*, 1998, Legierse, 1998). The time dependency of effect concentration is predicted by the pharmacological model of "target occupation" of Legierse (1998). In addition, lethal time is proposed to increase with body size (Stuijfzand, 1999). The most important implication of this is that the impact of pesticides on populations might depend on the season of occurrence. In addition, the recovery of organisms depends on the individual age or size as well as on the length of the life cycle (van den Brink *et al.*, 1996).

The time parameter also is important in assessing impacts on aquatic species as demonstrated by Liess (1994) and Liess and Schulz (1996). Indeed, the authors showed that with the same short-term exposure of one hour it makes a big difference of how long effects were observed and whether or not sublethal endpoints were included. For example with 24h observation time, a caddisfly was found to have an LC50 of 23 µg/L fenvalerate, while an observation time of several months and a sublethal endpoint (development) revealed an effect at 0.1 µg/l.

Latitude may also lead to differences in threshold determination for chemicals, since it influences the voltinism in invertebrate species and subsequently the exposure profile of the most susceptible life-stages. From Partial Least Square regressions it seemed that all macroinvertebrate groups were more sensitive when the experiments were conducted at high latitudes (Mohlenberg *et al*., 2001). One of the reasons evoked is a lower number of generations per year in species of high latitudes. The highest sensitivity was obtained at high latitudes but outside the summer months.

The most comprehensive review on the issue is provided by Reinert *et al.* (2002). According to the authors, pulsed exposure is close to environmental exposure conditions, especially in low-order streams and in ponds adjacent to agricultural areas. The authors reviewed the current knowledge on impacts of time-varying or repeated exposures to chemicals in aquatic organisms. Conclusions of this work indicate that agrochemicals may be either more or less or equally toxic to aquatic organisms when exposures are repeated compared to single peak exposure, for equal daily mean concentrations. Indeed, a decrease in the sensitivity of organisms may be observed if the first pulse selects more robust individuals. On the other hand, cumulative effects occur when the first pulse weakens the organism. Effects may finally be similar compared to those induced by a single pulse if impacts of the pulses are independent. Time to onset of effects, time to reversibility of effects and recovery time were identified as keys parameters conditioning the reaction of species. It was then suggested to define a threshold concentration (e.g. being 10% of EC_{50} for acute effects) to be compared with peak concentrations within time, and accounting for intervals between peaks, compared with the time necessary to recovery. The determination of “Critical Body Residue” (CBR), being the amount of body residue responsible for toxic effects is recommended, either experimentally, or using simple calculation (e.g. an approach of CBR would be LC_{50} or NOEC multiplied by the BioConcentration Factor-BCF), as BCF clearly depends on hydrophobicity of the chemical (Deneer *et al*., 1999 in Reinert *et al*., 2002). However, the main limit of the use of CBR in risk assessment is its dependence on age, sex, fat content and other variables of an individual within a species.

Other authors have attempted to deal with time-varying exposure through the use of toxicokinetic models such as the DEBtox model of Kooijman and Bedaux (1996). The DEBtox model allows to express survival probability as a function of internal concentration of the toxicant, derived from external concentration and bio-concentration factor. The derived model requires as additional data the time course of the external concentration of toxicant (Péry et al., 2001). It has been successfully used to predict No Effects Concentration of organic and inorganic compounds in several invertebrate species, including *C. riparius* (Péry et al, 2001 and 2003).

Based on this synthesis, the authors recommended that time-varying exposure is considered if exposure profiles and chemical behaviour suggest pulsed scenarios, if the application interval, compound half-life and/or exposure modelling indicate that such exposure may occur, as it contributes to reduce uncertainty. A Risk Assessment tool to evaluate Duration and Recovery (RADAR) and a corresponding decision matrix for time-varying exposure is proposed Reinert *et al.* (2002).

As another means to reduce uncertainty in risk assessment using toxicological endpoints and PEC that correspond to different duration, Bonnomet *et al.* (2002) developed a mechanistic model to express LC_{50} as a function of time. Entry parameters for their model are effect concentration curve, a time constant and an effect velocity that may be derived from a classical data set.

Finally, the age of the contamination might also influence the toxicity of chemicals to aquatic organisms. Kraaij *et al.* (2002) showed that the bioavailability of a toxicant could vary greatly with time, with a tendency to a decrease in bioavailability with increasing contact time between sediment and hydrophobic organic compounds.

3.3.3.5 Life stages

Regarding the presence of certain life stages at certain periods, a comprehensive overview is lacking. Data are available for individual organisms or organism groups. At Wageningen, an overview for the life cycle strategies of invertebrates present in the clay ditches and in the experimental ditches is being prepared, and will be published in the course of 2004.

It is often argued that young stages of insects are more sensitive to chemicals than older ones (Stuijfzand, 1999, van der Geest, 2000). This is one of the reasons that young life stages are generally used in laboratory toxicity studies. Susceptibility of invertebrates generally decreases with maturity in some stoneflies (Sanders and Cope, 1968) and in four species of malacostracan crustaceans (Sanders, 1972). Second and third instars of *Limnephilus lunatus* were respectively two and one order of magnitude more sensitive than fifth instars to fenvalerate, based on effects on emergence success and temporal emergence pattern (Schulz and Liess, 2001). In midges (*Paratanytarsus parthenogeneticus*) exposed during all their development to acenaphtene and 2,4,6 trichlorophenol, it appeared that larval developments after hatching were the most sensitive stages (Meier *et al.*, 2000). A similar trend was observed in the midge *Chironomus riparius* exposed to methiocarb. Sensitivity, based on acute effects was found to be higher in second larval instar than in third and fourth ones (Péry et al., 2003). Partial least square regression also indicates that copepod nauplii would be on average 10% more sensitive than the adult population (Mohlenberg *et al.*, 2001).

The opposite tendency has been described for different larval stages of *Xaenopus laevi*, with metamorphic stages being about 30 times more sensitive than premetamorphic (Richards and Kendall, 2002). The authors explained this difference by the development stage of the enzymatic system, being able to generate metabolites being more toxic than the active substance, and by the absence of a protective envelope in premetaphorphs.

The relationship between development stages and sensitivity to toxicants may depend on the toxicant as shown by Hardersen and Wratten (2000). In *Xanthocnemis zealandica* (Odonata: Zygoptera), the sensitivity of aquatic life stages did not correlate with aquatic life-stage for azinphos methyl but did for carbaryl, most sensitive stages being the younger ones. In *Paracentrotus lividus* (Echinoderma: Echinodea), sperm cells appeared to be less sensitive to organotin compounds than embryos, and the relative toxicity of several organotin compounds for each stage was different (Arizzi Novelli *et al*., 2002). It was suggested that the toxicological properties of tin compounds clearly depend on the nature of the substituent (Mediterranean Action Plan, 1989, in Arizzi Novelli *et al.*, 2002).

It is also known for invertebrates that beyond or in addition to the life-stage, susceptibility greatly depends on the diet as enzymes that metabolise absorbed chemicals may be the same as enzymes involved in nutrient metabolism (Croft, 1975 and 1989). It has been demonstrated that differences between first and last instars of one species could often be much bigger than differences between species, which could be explained by a difference in enzyme activity (Stuijfzand, 1999). In consequence the ranking of species should be strongly dependant on the developmental stage (Stuijfzand, 1999).

The nature of sensitive endpoints may vary with the larval stage, as shown in the midge *C. riparius* exposed to HCBP (Hwang *et al.*, 2001). This raises the question of the definition of the parameters on which sensitivity threshold should be based for each life-stage before any sensitivity comparison is made.

Some information is also available for fish. In *Oryzias latipes*, effects induced if exposure to thiobencarb occurred for blastula or at initiation of heart beat were different (Villalobos *et al.*, 2000). In *Melanotaenia fluviatilis*, a decrease of toxicity of esfenvalerate was observed with increasing age, which was attributed to a decrease in pesticide uptake on a weight basis (Barry *et al.*, 1995).

As far as algae are concerned, “developmental stages” were found to influence sensitivity to pesticides. Kent and Currie (1995) showed that total cellular lipids in several species of algae are correlated with sensitivity to lipophilic pesticides. In particular, quantities of lipid free sterol and sterol ester composition of cellular membranes may be very different and clearly influence the sensitivity to fungicides, even in the same genus (Tuckey *et al*., 2002).

3.3.3.6 Biotic factors in the risk assessment

The influence of food on toxicant effects on aquatic communities is poorly covered in the literature. Nevertheless, observations match those already described in Section 3.3.1, i.e. that food may act by conditioning population dynamics but also by influencing the bioavailability of substances and their absorption by organisms. However, information about ecological parameters like food, but also predation and competition, is too limited to raise general conclusions on their relative importance in the global effect of a toxicant on organisms.

More information is available on factors like frequency and exposure duration. As concluded by Reinert *et al.* (2002), it may be of great interest to account for exposure variation within time, in particular if the expected variations are relevant as regards the species on which the risk assessment is performed (e.g. reversibility of effects or recovery time may take place). In these cases, the proposal to use CBR (Critical Body Residue) or time-expressed LC_{50} should be discussed, based on the output of current standard tests. If the use of a CBR is to be envisaged, then information on the variability of this parameter within a species should be established, at least for standard test species. Approaches such as toxicokinetic and toxicodynamic models (e.g. DEBtox of Kooijman and Bedaux (1996)) need to be further developed to help solve the question of varying exposure with time (see also section 3.1).

Available data on life-stage related responses to toxicants might also be interpreted according to the exposure profile. This raises the question of the duration of a time-window that could be used for the risk assessment. According to Stuijfzand (1999), the impact of pesticides on populations might depend on the season of contamination occurrence. Indeed the proportion of the different life-stages are season-related so that the season in which most sensitive life-stages are abundant may or not correspond to the “high pesticide peak(s)” season. In invertebrates, early life-stages may be very abundant in spring but it may not be the case for all species. This raises the question of an assessment based on a time-window being shorter than a season. For the time being, available data seem too scarce to bring a satisfactory conclusion. Nevertheless, it seems reasonable that for the most sensitive trophic level, risk assessment is based on species presenting the lowest voltinism. In this sense, the definition of short time-windows would not be necessary.

According to Forbes *et al.* (2001), based on analytical and simulation findings, population-level effects should not be greater than effects on individual-level endpoints for populations with multiplication rates close to one. The comparison of PNEC based on NOEC for juvenile survival in the laboratory and output of simulations showed that the current extrapolation was rather protective but situations were identified where it would not be the case. The authors suggested accounting for the relative frequency of different life cycle types and proportions of sensitive and insensitive taxonomic groups in communities, and the role of density-dependent influences on population dynamics.

An important task could then be to get a more accurate estimation of the proportion of species for which early-life stages are also the most sensitive ones. The literature review proposed above highlights the size, enzymatic maturity and morphology of organisms as parameters that greatly condition organisms' sensitivity. Since size increases in general during the development of a species, the two last parameters may need to be more comprehensively addressed.

It is concluded that biotic factors can be important influences on the toxicity of pesticides. However, there are currently insufficient data to develop general rules, and further research into these issues is needed.

3.4 Ecological factors that influence exposure

Many factors of physical but also biological nature may influence the exposure of aquatic species in a water body. Abiotic factors such as pH, oxygen concentration, organic or inorganic matter have been seen to play a role in the overall response of organisms to toxicants by conditioning the fate of a substance. As far as the resulting effects were concerned, it appeared that generalization as difficult to establish as these factors may interact through their influence on population dynamics and further on species assemblages. Biotic factors may also greatly influence the overall fate of substances and perhaps aquatic macrophytes provide the best illustration of this fact (see for example Crum et al., 1999; Hand et al., 2001).

These factors might be categorised either based on their nature (abiotic, biotic) or on the mechanism by which they influence the exposure of species:

- fate of active substances in the water column and/or the sediment, through dissipation processes;
- bioavailability of active substance through transformation processes or via adsorption/absorption processes;
- structuring action through the creation of new habitats, refuges, thus influencing exposure through the behaviour of aquatic species.

The two first points have already been reviewed under Section 3.3. The section below proposes to illustrate many aspects of exposure conditions in water bodies as influenced by both physical and biological descriptors of a water body itself together with its neighbouring environment, via the example of riparian areas.

3.4.1 Role of riparian vegetation in landscape management

Several authors have proposed riparian corridors as important structures in landscape management (see Maridet, 1995 and Piégay et al., 2003, for reviews). These reviews bring an important contribution in understanding the role of riparian structures as regulators of water bodies' quality, and have already been used in the frame of a regional management of riparian vegetation in France. These reviews are summarised below.

3.4.1.1 Influence on the input of matter and nutrients

Vegetated riparian zones exert both a direct and indirect influence on the dynamic of nutrients in streams. The direct effect consists in a filtration of sedimentation particles, in particular fine particles that are related to anthropogenic activities. Fine sediments may indeed have negative effects on fish gills respiration, but may also progressively fill in invertebrate's habitats. Finally it may reduce the oxygen concentration inside sediments. Vegetal structures are also known to reduce the run-off of nitrate and phosphates in water. The main plants involved in the filtration process are herbaceous and bushy plants. These plants constitute a "cover" that increases soil roughness and decreases run off velocity, and that lead to sedimentation of particles. Some indications about the percentage of nutrients that may be absorbed by a riparian zone are given in Table 3.13.

Table 3.13. Some indications about the percentage of nutrients that may be absorbed by a riparian zone.

Ref.	Localisation	Soil type	Slope (%)	Width (m)	Plants	Nutrients and sediments	% deduced
Peterjohn and Correl, 1984 Correl and Weller, 1989	Maryland (USA)	Clayey sand	5.44	19	Trees	NO_3-N NH_4-N PO_4-P SO_4	80-97 73 0 25
Dolye *et al*., 1977	Maryland (USA)	-	--	3.8 4	Trees Grass	Soluble N Soluble P Soluble P	94.7 99.7 62
Pinay and Decamps, 1988	Garonne (France)	Clay loam	-	30	Trees	NO_3-N	100
Jacobs and Gilliam (1985)	North Carolina (USA)	-	-	30	Trees	NO_3-N	> 99
Smith (1989)	New Zealand	Clayey loam	6-11	10-30	Grass	Total N and P Sediment	70-78 60-98
Neibling and Alberts (1979)	Michigan (USA)	-	7	6.1	Grass	Sediment	90

3.4.1.2 Influence on habitat structure

This effect is related to the production of woody fragments that reach the water bottom and provide habitats for several species (e.g. Dryopidae or Psychomyiidae larvae) and refuges, and may lead to a more diverse habitat for fish. Corridors containing trees seem more efficient in producing these fragments than other plants, because of their root system or through overhanging vegetation. These structures also provide physical refuges and allow prey organisms to avoid predation. Many species among fish prefer shaded areas when not feeding or foraging. Finally it is suggested that the presence of fragments could allow the coexistence of various species and various life-stages of these species, since it provides spawning refuges in the same area. Riparian corridors constitute sanctuaries for species associated with this type of ecosystem. These zones finally improve the connectivity between refuges in the landscape.

3.4.1.3 Influence on conditioning factors

Riparian vegetation also influencse the trophic functioning of the ecosystem by regulating solar radiation of surface water. This role is to be considered together with the dynamic of the water[18], and should be a benefit only in highly dynamic systems. The production of woody fragments constitutes food for invertebrates. Indications on the relative abundance of trophic groups among invertebrates of the sediment are proposed in Table 3.14.

Table 3.14. Indications of relative abundance of trophic groups among invertebrates of sediment.

Riparian zone	Insolation index (%)	Chlorophyll a (mg/m^2)	Trophic groups	Relative abundance (%)
Trees with high overhanging surface	11%	31.8 ± 4.5	Predators	6
			Herbivorous	0
			Shredders	16
			Filter feeders	7
			Collectors	32
			Gougers	39
Miscellaneous	28%	49.6 ± 7.7	Predators	11
			Herbivorous	0%
			Shredders	16
			Filter feeders	4
			Collectors	33
			Gougers	36
Grass	75%	83.9 ± 12.4	Predators	9
			Herbivorous	2
			Shredders	7
			Filter feeders	7
			Collectors	14
			Gougers	61

[18] The limit value for the “potential specific energy” between highly dynamic and slighly dynamic water bodies is proposed to be 35 W/m^2.

Riparian vegetation also influences the temperature of water. Variations of maximum temperature may reach 3-10°C in summer for streams without any riparian cover, whereas variations in minimum temperatures are less significant (1-2°C) in winter. Changes in temperature conditions influence aquatic communities both directly (development cycle) and indirectly (food quality, oxygen solubility in water).

As a summary of this synthesis, an index of perturbation of the global functioning in aquatic ecosystems as a consequence of a drastic modification of riparian areas was proposed (see Table 3.15).

Table 3..15. Incidence of modification of riparian zones.

Water body classification			Perturbation induced (out of 20)			
Geology of basin	Climatic limit (T℃)	Water body	Excess in primary producers	Critical temperature reached (19℃)	Aquatic habitat modified	Global functioning (out of 60)
Limestone sedimentary massif	≥ 20	Low energy	20	20	20	60
		Low energy Moving sediment	7	8	15	30
		High energy Rocky sediment	10	6	12	28
	< 20	Low energy	18	5	20	43
		Low energy Moving sediment	5	0	15	20
		High energy Rocky sediment	8	0	12	20
Crystalline massif	≥ 20	Low energy	10	20	20	50
		Low energy Moving sediment	2	8	15	25
		High energy Rocky sediment	3	6	12	21
	< 20	Low energy	9	5	20	34
		Low energy Moving sediment	1	0	15	16
		High energy Rocky sediment	2	0	12	14

3.4.2 Conclusions

Riparian vegetation is often considered to be a filtering constituent of the landscape as it may (i) limit drift and protect the water body from aerial contamination (see e.g. De Snoo, 2001), (ii) limit run-off mediated transfers. From this review, other roles may be included such as

(iii) influence of the fate of active substances through temperature and solar radiation (iv) influence on both fate and impact through a global impact on general functioning and (v) influence on recovery potential. The biological and ecological aspects need to be more precisely documented before their use in risk assessment. In contrast, fate aspects seem more ready to use.

This example then completes the most known case of aquatic macrophytes, which often comprise a significant component of the biomass in aquatic ecosystems, from both a structural perspective (in relation to providing food sources and substrates for organisms) and fate aspects (in relation to a substantial influence on the dissipation and degradation of pesticides, e.g. in Crum et al. (1999) and Hand et al. (2001)).

In general, research is still needed to better describe the influence of physical descriptors of water bodies (e.g. presence of macrophytes, presence of riparian vegetation, organic matter charge) on the fate and bioavailability of chemicals, and to develop ways to model this.

As far as the influence of such factors on assemblage is concerned, it appears that many of them are already taken into account in the models having been developed to predict communities as it is the case for temperature (related to hydrology, climate and solar radiation), saprobic value (i.e. nutrient/oxygen status), trophic degree, microhabitats, etc. (see e.g. Tachet et al. (2000) for a review). Perhaps one key point for future research and in particular in model development will be to describe any of these factors for their influence on both sides (presence/absence of a species in assemblages and exposure) of the overall impact observed.

3.5 Factors that influence effects and recovery at the landscape level

Effects of toxicants and recovery of populations and communities at the landscape level differ in several aspects from the effects seen in laboratory and higher tier test systems. To a great extent, the causes are environmental variables and recovery processes that alter the effects seen in test systems and in the field. For example, Sherratt *et al*. (1999) demonstrated that acute effects are mainly driven by the sensitivity of species to the toxicant, whereas long term effects are also influenced (reduced) by the ability of species for recovery. In agricultural streams, Liess and Schulz (1999) demonstrated that (i) internal recovery occurred for most species within the year where the contamination occurred, and (ii) some species recovered due to external recovery by the following year. These aspects are reviewed below, together with input from available field and monitoring studies.

3.5.1 Effects at the community level

Effect monitoring studies are needed to determine whether effects are occurring in the field and if these effects are linked to pesticide usage. Such investigations have been successfully conducted in some cases, although establishing this causality may be difficult and the use of additional experimentation (e.g., in situ bioassays or mesocosms) can help to validate monitoring data. However , biological monitoring studies are useful for investigating the effects of multiple stressors as organisms will integrate these effects. It is important to identify whether it is possible to discriminate the impacts of individual stressors. Indeed, the presence of confounding factors and the natural variability of biological systems are a major problem in environmental monitoring as the effects due to pesticides and their magnitude may be difficult to detect.

It is unlikely that acute effects will be observed to a great extent as acute effects can be absent due to the occurrence of (chronic) development of tolerance at different levels of biological organisation (individuals, populations (common problem in pests), communities (e.g. Pollution Induced Community Tolerance, PICT, for examples see Bérard and Benninghoff (2001), Soldo and Behra (2000) and Seguin et al (2002)). Thus chronic effects based on the development of tolerance need to be taken in account.

The potential of environmental parameters to alter sensitivity of community needs to be defined. This leads to further questions such as the problem of geographical differences.

Annual temperature will lead to characteristic communities which may differ in sensitivity of species or may differ in ecological traits (e.g. generation time) which determine the duration of recovery. Also the stability of ecosystems (drying versus permanent water bodies) in relation to the variations of biological response to pesticides have not yet been investigated.

3.5.2 Recovery

Recovery of aquatic systems is defined in both HARAP (Campbell et al., 1999) and CLASSIC (Giddings et al., 2002) as the capacity of a disturbed system to reach a status being "comparable" to the status it had before disturbance occurred and within the range of an untreated system. Within the system, both populations, communities and functions of the systems are then subject to recovery. In this sense, "recovery" corresponds to one aspect of the "ecological resilience" defined by Gunderson (2000) in a recent review. Resilience is commonly described as (i) "the time required for a system to return to an equilibrium or steady state following a perturbation" or (ii) "the magnitude of disturbance that can be absorbed by a system before the system redefines its structure by changing variable and processes" (see also Carpenter and Cottingham, 1997). The main difference between the two definitions relies in the number (respectively one, or several) of stable states that is (are) assumed to exist for the system. According to Gunderson, the concept of "ecological resilience" presumes the existence of multiple stability domains and the tolerance of the system to perturbations that facilitate transition among stable states. Ecological resilience thus refers to the width or limit of a stability domain and is defined by the magnitude of disturbance that a system can absorb before it changes stable states.

This definition of "ecological resilience" is then far more comprehensive than the most dicotomous aspect (i.e. disturbance and recovery to the starting status) currently discussed. Indeed it is recognised that there is no "single equilibrium status" for aquatic ecosystems but rather a natural tendency to evolve between different stable states (for a debate see Leo and Levin, 1997). This concept is however an ambitious goal that is rather to be addressed in monitoring studies and tools are being developed within this framework. This aspect of the issue is rather difficult to investigate with the common tools used in risk assessment. In this document, "resilience aims" should then be "to ensure that no premature change occur in the system because of chemical pressure". Resilience will then be considered within a system during the exposure period to pesticides, but will also be considered taking into account the system within its landscape context.

Recovery concerns populations, community and functions within the ecosystem. Ecosystem recovery thus depends on population recovery and functional redundancy. As far as the functions of the ecosystem are concerned, it is recognised that functional redundancy, which means that the functions of the system are preserved even if some effects in some organisms are affected by a stress, may occur in any system. It occurs when several species are able to perform the same critical functions (see Solomon and Sibley, 2002).

Population recovery relies on two phenomena: (i) the presence of resting propagules within the system and on the number of surviving individuals (internal recovery) and (ii) input from neighbouring systems (external recovery), this last phenomenon being related to the dispersal capacity of a species. According to Van den Brink et al. (1996), parameters such as abundance in the system, species dynamics, age structure, genetic diversity, annual mortality rate, degree of isolation (within the population at the landscape level) and dispersal capacity thus condition recovery capacity of a given species.

Whatever the mechanism (external or internal) involved in recovery, it is important to know what species are able to recover. In this frame, recovery/resilience are often debated in the light of biological conservation concepts. The relationship between resilience and biodiversity conservation appears to be quite complex. In his synthesis, Gunderson (2000) states that resilience depends on the diversity of keystone species, being “drivers” (Walker, 1992) in the resilience process, but also on the diversity of other species, being considered as “potential drivers” in the resilience process. Indeed, functional redundancy conservation in a system may simply depend on species that were not “drivers” of key functions in the stable states of the ecosystem (Leo and Levin, 1997). Rather than keystone species, Hess and King (2002) proposed to focus on “focal species” in addressing impacts and risk issues. Focal species respond to the concepts of both “keystone” and “umbrella” species, i.e. when identifying threatening processes responsible for species decline and selecting a suite of species, each of which is considered most sensitive to each of the threatening processes. The area of encounter, dispersal capacity, dependency toward a process, and dependency towards a resource were then identified as threatening processes. Based on these assumptions, a score method has been proposed in order to identify which species may be focal ones.

Based on this complex relationship it appears that even when focal species could be identified as species of primary interest as regard conservation purposes, a joint analysis of ecosystem function implies also taking other species into account in addressing recovery issues. Indeed functional redundancy requires a successful colonisation of aquatic systems by redundant species to occur. It has to be mentioned that if from a scientific point of view assesments may in some cases be restricted to focal species, from a legal point of view this is critical because all species must be protected.

3.5.2.1 Internal recovery

Internal recovery is particularly important in addressing risk assessment for isolated water bodies. In this case, resilience of a system mainly relies on the internal resources of the system. Internal recovery (i.e. recovery out of the population affected) has been proved to be an important factor compensating the effect of pesticides. Species with a short generation time are likely to be less affected by fluctuating concentrations of toxicants due to a high potential for internal recovery. This is also supported by the results of mesocosm investigations (Van den Brink et al., 1996) as well as by basic ecological knowledge on the effect of stressors (Begon et al., 1990).

Internal recovery will be of limited importance when the generation time of species is greater than the (annual) cycle of pesticide contamination, i.e. species with a generation time greater than one year. Hence, the life cycle of species will determine their sensitivity to pesticides in the field situation (Liess et al., 2001a).

Another source of recovery is propagules. Propagules consist of dormant stages with reduced metabolic activity and a high resistance to desiccation and temperature (Williams, 1987, in Bilton et al., 2001). Such stages occur in many arthropods but also sponges, bryozoans, macrophytes and algae. Vegetal propagules consist of dormant seeds being devoted either to overwinter or to stockage purposes (De Winton et al., 2000). Germination that follows overwintering depends on physicochemical parameters including temperature (degree-day threshold) and sediment quality (nutrients, oxygen concentration) (Spencer et al., 2000). Percent sprouting among four species of aquatic plants from a laboratory-reconstituted bank ranged from 17 to 100%, depending of geographical and thus climatic parameters. Re-vegetation of ephemeral floodplain wetlands of the Nile River in South Africa has been studied for the germination success of sampled seed banks (Brock and Rogers, 1998). Sediments collected after flooding and germination events contained up to ca. 1400 individuals/m^2 (16 taxa). Persistent seed banks are constituted of seeds remaining in dormancy for a longer time window (De Winton et al., 2000).

Animal propagules consist in diapausing eggs of aquatic zooplankton, also being able to remain viable for decades or even over centuries (Weider et al., 1997). Both cyclical parthenogenic invertebrates, obligate parthenogenic invertebrates and obligate sexual invertebrates are known to build up resting propagule banks (De Meester et al., 2002). As an example, *Daphnia* species produce eggs that are protected into a highly resistant ephippium. Banks may contain as many as 1000-220 000 eggs/m^2 as observed for anacostracans in a south African pond (Brendonck and Riddoch, 1999).

Resting propagules are considered to play a role that is as important as nutrient content in spring plankton blooms observed in aquatic ecosystems (Boero et al., 1996). Activation mechanisms of the benthos and the importance of benthos-plankton relationship in the annual dynamic of aquatic species has been studied. It has been suggested that resting stages might constitute a potential biodiversity being much higher than the "realised" biodiversity formed by active organisms (Boero et al., 1996).

Few studies have been designed in order to investigate impacts of chemicals on overwintering resting stages of ponds. In a recent study, Henry et al. (2002) observed a clear regression of an aquatic channel from a eutrophic status to a mesotrophic status, occurring after a restoration action, through a colonisation by plants and a change in vegetation composition as the nutrient level decreased after restoration. This was possible by the development of plant propagules being present in the sediment of water and through the connectivity with other water streams. In a recent study, Kreutzweiser et al. (2002) have observed prolonged effects of a neem extract on some copepod species resulting in continued reduced abundance in samplings performed 350 days post-treatment. These last results raise the question of the susceptibility of resting stages inside egg banks to chemicals, despite the fact that they are supposed to be broadly protected against chemical stress through their location inside the sediment layer.

The importance of sediment in the aquatic systems' capacity to recover has been highlighted by the review of Brouver et al. (2002). It is concluded from this review that the recovery capacity of softwater lakes strongly (as regards macrophyte vegetation) depends on whether or not the sediment layer also was affected by chemical modifications, in addition to the water column. Consequently, short-term recovery is scarcely observed and thus considered as unlikely when the sediment is also affected.

3.5.2.2 External recovery potential

External recovery may contribute significantly to the overall recovery capacity for some aquatic species, even at the field scale. But most of all, external recovery becomes a critical point in addressing risk assessment at the landscape level.

External recovery mainly consists of input into the system of organisms coming from other systems. In a recent review, Bilton et al. (2001) proposed to use as a definition for dispersal "the outward spreading of organisms or propagules from their point of origin or release; one-way movement of organisms from one home site to another", in accordance with the definition of Lincoln et al. (1998 in Bilton et al., 2001).

Evidence of high dispersal capacity within both animal and aquatic macrophytes has been recently reviewed by De Meester et al. (2002). Insight into dispersal knowledge has been provided by ecological research jointly with molecular (genetic) approaches. The high dispersal capacity of aquatic organisms is thought to be the underlying process of the commonly observed quick colonisation of "empty" aquatic habitats such as newly formed water bodies. For agricultural streams, the importance of less contaminated stream sections for external recovery was emphasised by Liess & Ohe (submitted). Dispersal is either active e.g. for aquatic insects, or passive through vectors such as wind, water and animals.

Active dispersal

Active dispersal refers to dispersal of organisms without any vector. It refers to the flight of adult insects that allows them to move from one pond to another. Insects show varying degrees of dispersal according to taxonomic group, situation and environmental conditions (Bilton et al., 2001).

Very few studies investigated active dispersal in outdoor ponds. External recovery through oviposition has been suggested in Chaoborus flavicans, in order to explain the continuous increase in the number of larvae in a pond (Peither et al., 1996). An example of oviposition-mediated recovery in Chaoborus americanus in ponds treated with the insecticide temephos is mentioned by Helgen et al. (1988, in Peither et al., 1996). Examples of active recolonization of aquatic systems through the deposition of eggs by semi-aquatic species following a chemical stress have been provided (Woin, 1998). In this study, recolonization concerned uni- and multivoltine species, predominantly mayflies and dipterans. A full recolonization was observed within 3 years. Among trichopterans, recovery was observed at the group level but the species composition was changed. These results raise the question of the time window that is necessary for an external recolonization to be efficient at a group level (in order to preserve representative species of a trophic level). This time window may be highly variable and is rather to be defined on a case by case basis.

Passive dispersal

Transport via wind or animals involves mainly resting propagules. In the case of water connectivity between ponds, transport may also concern individuals of active populations. Both animal and vegetal propagules may be water-dispersed. The seeds of many species are adapted to float at the water surface, by wearing aeriferous structures, pulpy arils or hydrophobic structures (Santamaria, 2002). Wind dispersion mainly concerns recently formed propagules, as most propagules usually appear first on the water surface and then diffuse towards sediment where resting propagule banks are formed (Charalambidou and Santamaria, 2002). The propensity to be wind-dispersed in relation with morphological traits is poorly described. Plant seeds may often wear “plumes” or hair that increase air resistance, contrary to invertebrate eggs (Brendonck and Riddoch, 1999). Egg-shell structuring may also lead to increased adherence to substrates thus limiting lifts via the air. However, anacostracan eggs were found to be wind-dispersed over short distances (Brendonck and Riddoch, 1999). Bryozoan cells may also be gas-filled, thus presenting buoyancy that enables them to be transported by wind or waterbirds (Bilton et al., 2001).

Animal-mediated dispersal is also described in the literature as “biotic connectivity” and refers to the movements of the biota present in wetlands (Amezaga et al., 2002). It mainly concerns resting stages of both animal and vegetal species. Animal-mediated transport of propagules may be external (i.e. out of the vector). In this case propagules remain externally attached to animals, e.g. to the feathers of birds. The chances of success in external transport seem to be related to the use by waterbirds of the habitat where adherent propagules are abundant (Green et al., 2002). Both animals (e.g. Gammarus) and vegetal propagules may be moved by external transport (Clausen et al, 2002, Green et al., 2002). Animal-mediated transport of propagules may also be internal (i.e. inside the vector, also called endozoochorous dispersal). Waterbirds may be efficient vectors for the transport of propagules (Clausen et al., 2002, Santamaria and Klaasen, 2002). Propagules are transported in the gut of birds, thus suggesting that a proportion of propagules are not digested. Propagules are then ejected in the faeces of the bird at the new site. This transport concerns organisms of suitable size, ranging from ostracods to aquatic macrophytes (Clausen et al., 2002). According to Clausen et al. (2002), the main conditions for a successful transport of plant seeds by waterbirds are (i) the necessity that birds feed on these plants, (ii) the availability of seeds occurring simultaneously with movements of birds (for long distances), (iii) the necessity of birds to fly with the gut at least partly filled, (iv) a rapid movement of birds so that seeds survive the journey and (v) linkages between appropriate habitats. At the landscape scale, constraints relative to points (ii) and (iv) are less important. The conditions for successful dispersal of propagules are then mainly the presence of waterbirds feeding from a pond, survival of propagules in the digestive tract and excretion of propagules over another pond.

As an example, Anatidae have been demonstrated to be efficient seed transporters as they exploit highly seasonal food resources (Charalambidou and Santamaria, 2002, Clausen et al., 2002, Green et al., 2002). Both sedimentary and floating propagules are consumed and thus may be transferred from one pond to another (Green et al., 2002). Examples of birds studied for their contribution to animal and vegetal propagule dispersal is provided in Table 3.16. As far as plants are concerned, there are great differences in the survival chances of seed inside the gut of waterbirds (Charalambidou and Santamaria, 2002, Green et al., 2002). Examples of species influenced by waterbird-mediated dispersal are provided in Table 3.17.

For animals, many species of variable stages may be transported by waterbirds, since as is the case for plants, many of them constitute a food resource. Survival chances may also greatly vary from one species to another. Examples of species are provided in Table 3.18.

Table 3.16. Species used in propagule feeding experiments, from Charalambidou and Santamaria, 2002.

Waterbird species used in propagule feeding experiments:
Anas plathyrhynchos
Anas strepera
Anas acuta
Anas fulvicula
Aythya australis
Aythya fuligula
Fulica atra
Anser albifrons
Triga flavipes
Calidris minutilla
Nycticorax nycticorax
Phoenicopterus rubber
Tadorna tadorna

Table 3.17. Plant species concerned by dispersal through waterbirds.

Species	Viability (%)	Reference
Potamogeton sp.	0-55	Charalambidou and Santamaria, 2002
Ruppia		Clausen et al., 2002
Myriophyllum		Clausen et al., 2002
Zannichellia		Clausen et al., 2002
Zostera		Clausen et al., 2002
Chara		Clausen et al., 2002
Najas		Clausen et al., 2002
Wiesneria		Charalambidou and Santamaria, 2002
Echinochloa crusgalli	0	Charalambidou and Santamaria, 2002
Leptochloa fascicularis	7-57	Charalambidou and Santamaria, 2002
Nymphea alba		Charalambidou and Santamaria, 2002
Nuphar lutea		Charalambidou and Santamaria, 2002
Nastrurdium officinale	100	Charalambidou and Santamaria, 2002
Nymphoides peltata		Charalambidou and Santamaria, 2002
Polygonum bicorne	0.07	Charalambidou and Santamaria, 2002
Scirpus paludosus	36-50	Charalambidou and Santamaria, 2002

Table 3.18. Invertebrate species concerned by dispersal through animal and air vectors.

Species	Vector	Reference
Daphnia ephippia…………………………	Waterbird gut	Green et al., 2002
Larval and juvenile stages of zebra mussels…………………………………	mallard duck (external)	Bilton et al., 2002
Adults and juveniles of sphaeriid bivalves………………………………	insects and amphibians (external)	Bilton et al., 2002
Eggs and adult of limpet *(Patella)*………	insect (external)	Bilton et al., 2002
Juvenile pond snails (*Lymnaea stagnalis, Stagnicola elodes, Helisoma triviolis*)………………………………..		Bilton et al., 2002
Adult ostracoda………………………...	swans (external), air	
Adult amphipoda (*Hyalella azteca, Gammarus lacustris*)…………………..	notonecta (external), birds (gut)	Bilton et al., 2002
Adult and cocoons of ecotparasitic leeches……………………………….	beaver and muskrat (external)	Bilton et al., 2002
Larval water mites *(hydracarina)*……….	mallard duck (external)	Bilton et al., 2002
Crustaceans eggs……………………….	insects (external)	Bilton et al., 2002
Eggs of brine shrimp *(Artemia salina)*…	ducks (digestive tract)	Bilton et al., 2002
Statoblast of freshwater bryozoans…….	animals (digestive tract)	Bilton et al., 2002
	ducks, amphibians (digestive tract)	Bilton et al., 2002

Dispersal may occur at very different geographical scales, i.e. local, regional and also continental. Studies on the distribution of multilocus genotypes in parthenogenetic species such as *Daphnia* sp. (Weider and Hobaek, 1997, Weider et al., 1996, 1999b, in De Meester et al., 2002) or bryozoans (Freeland et al., 2000 in De Meester et al., 2002) showed that the distances covered could reach 1000 km. It is recognised that waterbirds are effective at performing long distance dispersal of aquatic organisms, as they are often migratory birds (Green *et al*., 2002, Clausen et al., 2002). Long distance (100-20000 m) transport has often been observed for terrestrial plants, but is supposed to be rather unusual for aquatic organisms, which are usually subject to shorter dispersal (0-100 m) (Clausen et al., 2002). As an example, ducks and geese often fly about ten kilometres between feeding and roosting sites (Tamisier and Dehorter, 1999 in Green et al., 2002). At a local scale, bird-mediated passive transport of plant seeds, both internal and external, is thought to be frequent (Santamaria, 2002) and a key aspect determining wetland connectedness even in the absence of direct hydrologic links (Amezaga et al., 2002).

3.5.2.3 Recovery and risk assessment

As stated above, the integration of either internal or external recovery in ecological risk assessment may in principle be performed at both the field and landscape scales. Over recent years, several models for predicting internal recovery rates have been developed, and methods for quantifying population recovery rates have recently been discussed by Barnthouse (2004). As an example the model of Waito et al. (2002) was developed in order to extrapolate data from single species tests to field ecosystems, taking into account growth rate together with fate aspects for species of zooplankton, benthic insects and invertebrates, omnivorous and piscivorous fish (see also Section 3.3.1.7). Another model has been proposed by Kuhn *et al.* (2002) based on the analysis of population growth elasticity. Both 10-d acute and 70-d reproduction tests were performed on the amphipod *Ampelisca abdita* with cadmium and acid volatile sulfide, and parameters such as cumulative survival as a function of age class and related growth rate were derived. The model was used to deduce impact of the chemicals on the population from single test studies.

A SETAC Pellston workshop in 2003 (http://www.setac.org/eraag/era_pop_pellston.htm) extensively reviewed population-level approaches in ecological risk assessment, including discussion and examples of potential modelling approaches (publication in preparation). Further useful information and guidance can be found in these publications.

In general, the use of modelling approaches should be considered on a case-by-case basis, and is not particularly amenable to specific guidance at this stage. In particular, population modelling requires life-history data for parameterization purposes. One approach that has been suggested to overcome this is the use of models with simplified life-history scenarios (Calow et al., 1997). Such an approach may constitute a useful first tier, especially for exploring those types of life history that may be vulnerable to a particular pesticide. Where reasonable amounts of life-history data are available, individual-based models can be developed for a specific organism in order to estimate recovery rates under a range of conditions. However, in the future in order to improve the potential for the use of population models, efforts should be made to collect life-history data. A number of projects are currently aiming to do this, including the FreshwaterLife project (www.Freshwaterlife.org) and the UK PSD WEBFRAM project.

The integration of internal recovery from resting stages into the risk assessment also implies to have a good prediction of the behaviour of chemicals in the sediment. The key point is: are resting stages susceptible to be exposed to chemicals and to which ones? It also implies that we know the sensitivity of resting stages compared to other ones (for the time being there is little information about that subject).

External recovery has so far received much less attention than internal recovery, although it is acknowledged to be an important process. The integration of external recovery implies to consider and quantify the capacity to disperse of species of concern. Some data are available for invertebrates, in particular for species presenting aerial adults thus able to disperse actively (see e.g. Tachet et al, 2000 for a review). But passive dispersal is far less easy to consider, while dispersal of phytoplankton and macrophytes is mainly a passive one. In the latter case, there will be differences between physically isolated and non- isolated water bodies.

Relatively little work has been carried out into the development of meta-population models, although the theoretical constructs have been developed (Wiens, 1997). There are also relatively few data concerning the dispersal of aquatic organisms through the landscape, although research interest in this area is growing (e.g. Conrad et al., 1999; Bilton et al., 2001; Purse et al., 2003). Such approaches may prove useful for landscape level assessments in the future, but further work is needed to develop practical tools.

3.5.3 Examples of landscape level studies

Landscape investigations of invertebrate communities in running waters show that species number and composition are largely dependent on the environmental factors to which the communities are exposed (Ruse, 2000; Wally and Fontana, 1998). They will comprise relatively few species if these factors are in a range considerably different from the optimum required for most of the species. In agricultural streams such factors include, for example, a highly dynamic discharge (Sheldon et al., 2002), siltation of the streambed (Vuori and Joensuu, 1996) and pesticide entry (Liess and Schulz, 1999). Therefore the question arises to what extent the observed variability in species number in the field is associated with pesticides or other stressors linked to agricultural activities.

The effect of pesticides in agricultural streams was demonstrated by the use of in situ bioassays (Baughman et al., 1989, Matthiesen et al., 1995) and by a combination of in situ bioassay and field investigation (Liess and Schulz, 1999). Nevertheless, the relative importance of pesticides and other factors on the community can be more easily studied when investigating several streams as multivariate statistics can identify the average importance of pesticide stress in determining community structure in a certain area.

Probst et. al. (in press) combined land use data (ATKIS) with monitoring results within a GIS. The authors estimated the stress originating from arable land by the factor “risk of runoff”, which was derived from a runoff-model (rainfall induced surface runoff). Multivariate analysis explained 39.9% of the variance in species number, revealing stream width as the most important factor (25.3%) followed by risk of runoff (9.7%). The results showed that wider streams – with or without agricultural stressors - contained significantly higher species numbers than smaller streams. This can be explained by potentially more diverse in-stream structures leading to more habitats and niches. However, negative effects on species number owing to runoff from arable land could be distinguished from the effect of stream width: the number of species within each stream width class significantly decreased with increasing risk of runoff. Therefore the factor “risk of runoff” is considered to express a significant proportion of the variability in macroinvertebrate communities caused by stressors of agricultural origin.

De Zwart (2003) used pesticide usage data in a GIS and combined this with models for exposure of surface water. The resulting exposure levels were compared to species sensitivity distributions and rules for mixture toxicity calculation and a risk estimate for the species assemblage in the aquatic ecosystem was made. The risk is expressed as the proportion of species likely to be suffering any effect from the exposure. The results show that 95% of the predicted risk is caused by only 7 of 261 pesticide ingredients, mainly used in potato crop. Comparable results were found by De Snoo (1999), who used an environmental yardstick and compared this with pesticide use. The maximum local risk was estimated to affect about 50% of the species. The results were then compared to field observations. Although some indications were gathered, the number of field observations was not sufficient to generate quantitative results.

From the field studies available it is concluded that studies at the landscape level are scarce, and results are variable. Nevertheless studies showing effects in the field are present, and can be used to validate modelling.

3.6 Conclusions

Risk assessment at Step 4 should not only consider exposure assessment, but all options for refinement, including ecological considerations. One important development in this area would be the definition of the ecological characteristics (biotic and abiotic) of the FOCUS surface water scenarios. Information of this sort could be used in the future to refine both the exposure and effects assessment. Although there is no off-the-shelf approach available for this at present, it is considered that sufficient data and models exist to at least allow this to be done at a low level of taxonomic resolution (e.g. family level).

Abiotic and biotic factors can have an influence on the toxicity of pesticides. In this context, standard studies aim at maximizing exposure of organisms in order to provide worst-case estimates of effect concentrations for the exposed stages of the tested species. Higher-tier studies and risk assessments might take into account how such factors may influence effects, but this needs to be done on a case-by-case basis due to compound specific differences in the interaction of toxicant and abiotic or biotic factors.

One of the challenges confronting risk assessors in light of the FOCUS surface water scenario developments is the time-varying exposure profile of concentration produced at Step 3, which can be at odds with the maintained exposure conditions in standard toxicity tests. A review of potential approaches for addressing this has been conducted.

Furthermore, moving to the landscape level provides opportunities for considering recovery potential, both internally (from within the water body of concern) and externally (from neighbouring waters). Potential approaches for developing these techniques have been reviewed.

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ANNEX 1 DATA LAYERS FOR INTEGRATED SPATIAL ANALYSES

Additional listing of data layers is available on the FOCUS web site at http://viso.ei.jrc.it/focus.

List of Elevation Data and Providers

COVERAGE	SOURCE	PRODUCT NAME	PRODUCT TYPE	SPACING	ACCURACY		COMMENT
				Horizontal	Horizontal	Vertical	
Austria	Bundesamt für Eich- und Vermessungswesen	DGM	Grid	50m		+/- 5m	4.21 per sqr. km
		DGM	grid	100m			2.11 per sqr. km
Belgium	Institut Geographique National	DTED Level 2	grid	30m			
		DTED Level 1	grid	90m			
Denmark	Kort & Matrikelstyrelsen	DHM 5m	contour	5m			
		DHM 50	grid	50m		+/- 2.5m	
		DHM 200	grid	200m			from 5m equidist. Contours.
Finland	National Landsurvey of Finland	DEM	grid	25m			Defense Force permission required
		DEM	grid	50m			1 - 5 user license.
		DEM	grid	200m			New product, older data available.
France	Institut Geographique National	BD ALTI	grid	75m			FF 1-40 per sqr. km, average of FF20 assumed
	Geomantics		grid	75m			$12000 per 1deg, no longer available
Germany	Bundesamt für Kartographie und Geodäsie	DHM/M745	grid	20x30m	+/- 26m	+/- 20m	Other products different for each Land
Greece	Hellenic Military Geographical Service	ELLAS1M	contour	20m			184.00 per 1:50.000 sheet, 50 map sheets estimated
Ireland	Suirbhéireacht Ordanáis na Éireann		grid	1000m			from GDDD
			contour	1000m			N.A.
Italy	Instituto Geographico Militario	DTM20	grid	20m			

COVERAGE	SOURCE	PRODUCT NAME	PRODUCT TYPE	SPACING	ACCURACY		COMMENT
				Horizontal	Horizontal	Vertical	
		DTED L2?		1"			480,000
		DTED L1?		3"			140,000
Luxembourg	Administration du Cadastre et de la Topographie						No information found
Netherlands	Topographic Service						Only topo information found
Portugal	Instituto Portugues de Cartografia e Cadastro (IGEOE)	DTED Level 1		100m	10m	+/- 2m	
		DTED Level 2		30m	10m	+/- 2m	
Spain	Centro Nacional de Informacion Geografica	DTM200		200m			ESP 207.00 per 1:100.000 map, 50 map sheets est.
Sweden	Lantmäteriverket	GSD-Hoejd-databanken	grid	50m		2.5m	
		GSD contours	contour	5m			
		GSD contours	contour	10-25m			
UK	Ordnance Survey	Panorama	grid	50m			Copyright stringent
European DEM	Eurostat GISCO	DEEU3M	grid	30 arc"		18m RMS	Mainly GTOPO30
	GAF, Geosys	MONA PRO*	grid	75m		3-4m	Sub-set available.
			grid	100m		From 75m	From 75m data.
			grid	250m		From 75m	From 75m data.
	Geo Strategies SA	DTM	grid	100m			Eastern Europe only.
	Intermap	Global Terrain	grid	5m		1m	Partial cover of EU MS.
	Nigel Press Associates	EuroDEM	grid	75m		18m	Limited European coverage.
			grid	250m		From 75m	Limited European coverage.
	Ten-to-Ten (Digitech)		grid	75m		?	Partial coverage of regions
	NIMA / USGS	GTOPO30	grid	30 arc"		18m RMS	Available through USGS
		DTED Level 1	grid	90m			Subject to permission from DoD.
		DTED Level 2	grid	30m			Subject to permission from DoD.

* Limited European coverage: AU, BE, DE, FR, GR (54%), IT, LU, NL (81%), PO, SP (in general below 56deg N).

List of Line Data Providers

Supplier	Product	Cover	Format	Storage	Scale	Price	Comments
SCALE 1:1,000,000 (or smaller)							
MapInfo	AA Automaps*	Europe	Vector	MapInfo	1:1,000,000	£ 2500	Price for all layers. £ 750 for open water bodies alone.
Bartholomew	Euromaps on CD	Europe	Vector	ARC/Info, Shape	1:1,000,000	£ 1100	20 layer, wide European coverage.
	European Database	Europe	Raster + vector	ARC/Info, Shape	1:1,000,000	£ 1245	More up-to-date, used in ERICA project.
EuroGeographics	EuroGlobalMaps	Europe	Vector		1:1,000,000		Availability pending
Eurostat GISCO	WPEU1M	Europe	Vector	ARC/Info	1:1,000,000		Mainly Bartholomew
ESRI	ARC World	World	Vector	ARC/Info	1:3,000,000	$ 395	Modified DCW data.
GeoComm	Free GIS Data	World	Vector		1:1,000,000	download	Links to DCW.
Global Insight Europa Technologies	Global Insight Plus	World	Vector	MapInfo, Shape	1:1,000,000	$ 2495	Advertised as replacement for DCW.
GRID, Arendahl	Baltic Sea	Baltic	Raster / Vector	Idrisi	1:1,000,000	download	Drainage basins defined for Baltic region.
USGS	DCW	World	Vector	various	1:1,000,000	download	Widely used base data.
	Hydro 1k	World	Raster / Vector	BIL, .E00	1:1,000,000	download	http://edcdaac.usgs.gov/gtopo30/hydro/readme.html
SCALE 1:250,000 (or larger)							
AND Mapping	European Global Road Database	Europe	Vector	Shape (through ESRI)	1:250,000	On request	Used in Arc Europe from ESRI.
ESRI	ARC Europe	Europe	Vector	Shape	1:250,000	$ 600**	Based on AND Data BV.
MapInfo	Cartique™ Mapping	Europe	Vector	MapInfo	1:300,000	£ 15500	27 layers, EU15+ coverage open water bodies not alone

Product no longer listed on MapInfo site.
** In USA, Euro 1,200 in Europe.

ANNEX 2 CONTACT INFORMATION FOR SPATIAL DATA LAYERS (EU & MEMBER STATE LEVEL)

This table is an extract from a version containing additional fields and tables, available on the FOCUS web site at http://viso.ei.jrc.it/focus.

Notes for use of the contact information table:

column	description	predefined categories / examples
Country/State	country in which the provider is located, or for which the coverage of data extends	e.g. Germany, Spain, USA, Queensland
State/county/administrative district	administrative unit to which the provider is assigned	e.g. Hessian, Loire Bretagne, Northern Ireland, Long Island
Provider type	classification of provider type into predefined classes	categories: authority, company, university/institution/research
Provider	official provider name	e.g. The Ordnance Survey Great Britain, University Bern
Web Site	world wide web address	

Country/State	Provider	State, County, Administrative District	Provider Type	Web Site
Europe	Eurostat		authority	http://europa.eu.int/comm/eurostat/
Europe	The European Soil Bureau		authority	http://esb.aris.sai.jrc.it/
Europe	Geo Strategies	Central & Eastern Europe	company	http://www.geo-strategies.com/home.htm
-	Multimap		company	http://www.nwl.ac.uk/ih/www/research/bhost.html
-	GeoStore		company	http://www.geostore.com/
-	Ionia		company	http://shark1.esrin.esa.it/ionia/EARTH/LST/AVHRR/welcome.html
-	Geo Community		company	http://search.geocomm.com/cgi-bin/texis/db/search/?db=gdd&query=Germany
Europe	Euromap		company	http://www.euromap.de/

Country/State	Provider	State, County, Administrative District	Provider Type	Web Site
-	Globeexplorer		company	http://www.globexplorer.com/
-	GAF AG		company	http://www.gaf.de/data.html
Europe	European Water Association		university/institution/research	http://www.ewpca.de/
Europe	Eurogeographics		university/institution/research	http://www.eurogeographics.org/
Europe	Euro-Mediterranean Information System on the Know-how in the Water sector		university/institution/research	http://www.emwis.org/
Europe	AND Mapping USA, Inc.		company	www.andusa.com
Europe	AND International Publishers		company	www.and.com
Europe	Bartholomew		company	www.bartholomewmaps.com
Europe	ESRI, Redlands, CA		company	http://www.esri.com/
Europe	European Environment Agency (EEA)		authority	http://www.eea.eu.int/
Europe	Joint Research Centre (JRC)		university/institution/research	http://www.jrc.cec.eu.int/welcome.htm
Europe	MDC – Environmental Satellite Data Centre		university/institution/research	http://directory.eoportal.org/info_MDCEnvironmentalSatelliteDataCentre.html
Europe	NATLAN			http://www.eionet.eu.int/Best_Practice/Acronyms/NATLAN
Europe	RISO National Laboratory		university/institution/research	http://www.risoe.dk/
Europe	MACON, USA		company	http://www.macon.de/en/partner/usa.htm
Europe	EuroGeographics		authority	http://www.eurogeographics.org/eng/01_about.asp
Europe	USGS		authority	http://www.usgs.gov/
Europe	Global Historical Climatology Network (GHCN)		authority	http://cdiac.esd.ornl.gov/ghcn/ghcn.html
Europe	United Nations Environment Programme (UNEP) Global Resource Information Database (GRID)		authority	http://www.grida.no/
Australia	Geoscience Australia		authority	http://www.auslig.gov.au/
Austria	Bundesamt für Eich- und Vermessungswesen		authority	http://www.bev.gv.at/prodinfo/kartographische_modelle/amap_3f.htm
Austria	Geologische Bundesanstalt		authority	http://www.geolba.ac.at/GBADB1/index.html
Austria	Federal Ministry for Land and Foresty, Environment and Water Conservation		authority	http://www.lebensministerium.at/home/

Country/State	Provider	State, County, Administrative District	Provider Type	Web Site
Austria	Federal Office and Research Center for Agriculture		authority	http://www7.bfl.at/kontakt/
Austria	Technische Uni Wien		university/institution/research	http://www.ipf.tuwien.ac.at/produkte/ezggew.html
Austria	Universität für Bodenkultur		university/institution/research	http://www.boku.ac.at/
Austria	Statistic Austria		university/institution/research	http://www.statistik.at/index.html
Belgium	Nationaal Geographisch Instituut		authority	http://www.ngi.be/FR/FR1-5-6.shtm
Belgium	Statistics Belgium		authority	http://statbel.fgov.be/
Belgium	Aquaterra		company	http://www.aquaterra.be/
Belgium	Institute of Nature Conservation		university/institution/research	http://www.instnat.be/navi_ineng.htm
Belgium	Biodiversity Resources Institute		university/institution/research	http://betula.br.fgov.be/BIODIV/
Belgium	Faculté Universitaire des Sciences Agronomiques de Gembloux		university/institution/research	http://www.fsagx.ac.be/ha/recent1.htm#Design%20Floods%20Assessment%20with%20a%20mathematical
Belgium	Agricultural Resarch Center Gent		university/institution/research	http://www.clo.fgov.be/
Belgium	Laboratory of Hydrology and water management		university/institution/research	http://taoren.rug.ac.be/
Belgium	Belgian Geology Resources		university/institution/research	http://users.skynet.be/Belgeol/
Belgium	GIS Vlaanderen		authority	http://www.mina.vlaanderen.be/
Belgium	Tele Atlas		company	http://www.teleatlas.com/landingpage.jsp
Bulgaria	National Statistic Institute		authority	http://www.nsi.bg/Index_e.htm
Bulgaria	Ministry of Regional Development and Public works		authority	http://www.mrrb.government.bg/inbrief.php.htm
Canada	Centre for Topographic Information		authority	http://maps.nrcan.gc.ca/
Canada	The Geodetic Survey Division		authority	http://www.geod.emr.ca/
Canada	The National Resources Canada		authority	http://www.aft.pfc.forestry.ca/
Canada	Intermap		company	http://www.intermap.ca/
Croatia	Croatian Bureau of Statistics		authority	http://www.dzs.hr/
Croatia	State Geodetic Administration		authority	http://www.dgu.tel.hr/dgu/index-eng.htm
Cyprus	Statistical Service		authority	http://www.pio.gov.cy/dsr/index.html
Cyprus	Cyprus Department of Lands and Surveys		authority	http://www.megrin.org/gddd/orgs/os_18.htm
Czech Republic	Ministry of the Environment		authority	http://www.env.cz/
Czech Republic	Czech Statistical Office		authority	http://www.czso.cz/

Country/State	Provider	State, County, Administrative District	Provider Type	Web Site
Czech Republic	Czech Office for Surveying, Mapping and Cadastre		authority	http://www.vugtk.cz/~cuzk/adr01_en/sortim_en.html
Czech Republic	Czech Office for Surveying, Mapping and Cadastre		authority	http://www.vugtk.cz/~cuzk/adr01_en/sortim_en.html
Czech Republic	GISAT		company	http://www.gisat.cz/ENG/index.php
Czech Republic	GISAT		company	http://www.gisat.cz/ENG/index.php
Denmark	Statistic Institute		authority	http://www.dst.dk/dst/dstframeset_1024_en.asp
Denmark	Geological Survey of Denmark and Greenland		authority	http://www.agrsci.dk/jbs/jordbund/index_uk.html
Denmark	National Environmental Research Institute		authority	http://www.dmu.dk/forside_en.asp
Denmark	Kampsax		company	http://www.geodata-info.dk/ds.asp?DS=278&LA=2
Denmark	Kort& Matrikelstyrelsen		company	http://www.kms.dk/index_en.html
Estonia	Statistic Institute		authority	http://www.stat.ee/
Estonia	Estonian Land Board		authority	http://www.maaamet.ee/yldinfo/aboutus.php
Finland	Statistic Institute		authority	http://tilastokeskus.fi/index_en.html
Finland	The National Land Survey of Finland		authority	http://www.nls.fi/kartta/maps/gis_eur7.html
France	Department for Agriculture and Development		authority	http://www.adasea.net/adasea/#
France	Agence de l'eau Adour Garonne	Adour Garonne	authority	http://www.eau-adour-garonne.fr/
France	Agence de l'eau Artois-Picardie	Artois-Picardie	authority	http://www.eau-artois-picardie.fr/index.htm
France	Loire Bretagne Water Agency	Loire Bretagne	authority	http://www.eau-loire-bretagne.fr/english/FRNTANGL.HTM
France	Agencede l'eau Rhin-Meuse	Rhin-Meuse	authority	http://www.eau-rhin-meuse.fr/
France	Agence de l'eau Rhône-Mediterranée-Corse	Rhône - Mediterranée-Corse	authority	http://www.eaufrance.tm.fr/
France	Agence de l'eau Seine -Normandie	Seine Normandie	authority	http://www.eau-seine-normandie.fr/
France	Statistic Institute, studies of economy		authority	http://www.insee.fr/en/home/home_page.asp
France	Ministry of Ecology and Development		authority	http://www.environnement.gouv.fr/
France	Agences de l'eau		authority	http://www.eaufrance.tm.fr/

Country/State	Provider	State, County, Administrative District	Provider Type	Web Site
France	Institut Géographique National		authority	http://www.ign.fr/fr/MP/produit/rasters/SCANdep/index.html
France	Institute National de la Recherche Agronomique		authority	http://www.inra.fr/cgi-bin/nph-engine/htdocs/USER/EDITIONS/index.mhtml?bidon=1028538539&gau=cartes.mhtml&langue=english
France	L´Europe vue du ciel		company	http://www.leuropevueduciel.com/
France	National Water- related Information and Documentation Service		university/institution/research	http://www.oieau.fr/anglais/fdocumentation.htm
France	French Water Data Network		university/institution/research	http://www.rnde.tm.fr/anglais/rnde.htm
France	CORINE landcover		university/institution/research	http://www.ifen.fr/pages/2corin.htm
France	GEOSYS		company	http://www.geosys.fr/english/sommaire/frsommai.htm
Germany	Landes Betrieb Vermessung Baden Württemberg	Baden Württemberg	authority	http://www.lv-bw.de/LVShop2/index.htm
Germany	Bayrische Landesvermessungsamt	Bayern	authority	www.geodaten.bayern.de
Germany	Landesvermessung und Geobasisinformation	Brandenburg	authority	http://www.lverma-bb.de/produkte.htm
Germany	Amt für Geoinformation und Vermessung	Hamburg	authority	http://www.hamburg.de/fhh/behoerden/behoerde_fuer_bau_und_verkehr/amt_fuer_geoinformation_und_vermessung/produkte.htm
Germany	Hessische Verwaltung für Regionalentwicklung Kataster und Flurneuordnung	Hessen	authority	http://www.hkvv.hessen.de/produkte/index.htm
Germany	Landesvermessungsamt Mecklenburg-Vorpommern	Mecklenburg-Vorpommern	authority	http://www.lverma-mv.de/
Germany	Vermessung- und Katasterverwaltung Niedersachsen	Niedersachsen	authority	http://www.vkv-ni.de/
Germany	Landesvermesungsamt Nordrhein-Westfahlen	Nordrhein-Westfalen	authority	http://www.lverma.nrw.de/produkte/framePRODUKTE.htm
Germany	Landesamt für Vermessung und Geobasis Information	Rheinland- Pfalz	authority	http://www.lvermgeo.rlp.de/menue_03.htm
Germany	Landesamt für Kataster-, Vermessungs- und Kartenwesen	Saarland	authority	http://www.lkvk.saarland.de/
Germany	Landesvermessungsamt Sachsen	Sachsen	authority	http://www.lverma.smi.sachsen.de/produkte/index_produkte.html

Country/State	Provider	State, County, Administrative District	Provider Type	Web Site
Germany	Vermessungs- und Katasterverwaltung Sachsen- Anhalt	Sachsen- Anhalt	authority	http://www.geobasis.sachsen-anhalt.de/
Germany	Landesvermessungsamt Schleswig-Holstein	Schleswig- Holstein	authority	http://www.schleswig-holstein.de/lverma/
Germany	Thüringer Kataster-und Vermessungsverwaltung	Thüringen	authority	http://www.thueringen.de/vermessung/Metadaten/index.htm
Germany	Statistisches Bundesamt		authority	http://www.destatis.de/
Germany	Aeroview		company	http://www.aeroview.de/
Germany	Bundesamt für Kartographie und Geodäsie (BKG)		authority	http://www.ifag.de/
Germany	Deutsches Fernerkundungsdatenzentrum (DFD)		authority	http://www.caf.dlr.de/caf/institut/dfd/
Germany	Gesellschaft für Angewandte Fernerkundung mbH (GAF)		company	http://www.gaf.de/
Greece	Statistic Institute		authority	http://www.statistics.gr/new_site/English/MainPage/index_eng.htm
Greece	Hellenic Mapping & Cadastral Organization		authority	http://www.okxe.gr/proioda/aerialphotography/index.html
Greece	ERAnet		company	http://www.eranet.gr/geodata/en/index.html
Greece	Hellenic Military Geographical Service		authority	http://www.gys.gr/english/EN1.htm
Hungary	Central Statistics Office		authority	http://www.ksh.hu/pls/ksh/docs/index_eng.html
Hungary	Institute of Geodesy, Cartography and Remote Sensing (FÖMI)		authority	http://fish.fomi.hu/angolfish/
Hungary	Netherlands Institute of Applied Geoscience		university/institution/research	http://www.nitg.tno.nl/eng/appl/g_resources/groundwater/514.shtml
Iceland	The National Land survey of Iceland		authority	http://www.lmi.is/landsurvey.nsf/pages/index.html
Iceland	Landlýsing		authority	http://www.lmi.is/landlysing/
Iceland	The National Energy Authority		authority	http://www.os.is/english/
Iceland	Agricultural Research Institute		authority	http://www.rala.is/
Iceland	Statistics Iceland		authority	http://www.hagstofa.is/
Ireland	Ordance Survey Ireland		authority	http://www.osi.ie/mapping/digital/index.shtml
Ireland	Central Statistics Office		authority	http://www.cso.ie/
Italy	Statistic Office		authority	http://www.istat.it/English/index.htm

Country/State	Provider	State, County, Administrative District	Provider Type	Web Site
Italy	National Environment Agency		authority	http://www.sinanet.anpa.it/
Italy	Italian Military Geographic Institute		authority	http://www.nettuno.it/fiera/igmi/uk_version/cata_uk.htm
Italy	The National Contact Point in Spain		university/institution/research	http://www.dstn.it/simn/semide/English/SEMIDE_eng.htm
Italy	National Water Research Institute		university/institution/research	http://www.irsa.rm.cnr.it/home.php
Lativa	National Mapping Organisation		authority	http://www.vzd.gov.lv/
Lativa	Statistic Office		authority	http://www.csb.lv/avidus.cfm
Lithunia	Statistic Office		authority	http://www.std.lt/default_e.htm
Lithunia	Institute for Geography		authority	http://www.geo.lt/
Lithunia	Institute for Geology		authority	http://www.geologin.lt/
Lithunia	Geological Survey of Lithunia		authority	http://www.lgt.lt/
Luxembourg	STATEC Statistic		authority	http://statec.gouvernement.lu/
Luxembourg	Administration du Cadastre et de la Topographie		authority	http://www.etat.lu/ACT/acceuil.html
Malta	National Statistic Institute		authority	http://www.gov.mt/frame.asp?l=2&url=http://www.nso.gov.mt/
Netherlands	Netherlands Institute of Applied Geoscience TNO- National Geological Survey		authority	http://www.nitg.tno.nl/eng/appl/general/214.shtml
Netherlands	Centre for Geoinformation		authority	http://flex082.girs.wau.nl/cgi/products/products.htm
Netherlands	Topografische Dienst Niederlande		authority	http://www.tdn.nl/
Netherlands	The Biology Department		authority	http://www.biol.rug.nl/
Netherlands	Atlas van Nederland		authority	http://avn.geog.uu.nl/
Netherlands	Institute for Biology		authority	http://www.nibi.nl/
Netherlands	Statistics Netherlands		authority	http://www.cbs.nl/en/
Netherlands	Agricultural Economics Research Institute		authority	http://www.lei.wageningen-ur.nl/
Netherlands	Ministry of Agriculture, Nature Management and Fisheries		authority	http://www.minlnv.nl/international/
Netherlands	DTB-nat		company	http://www.minvenw.nl/rws/mdi/geoloket/dtbnat.html
Netherlands	Wageningen UR		company	http://geodesk.girs.wau.nl/geokey/select.htm
Netherlands	Alterra		company	http://www.alterra.nl/english/

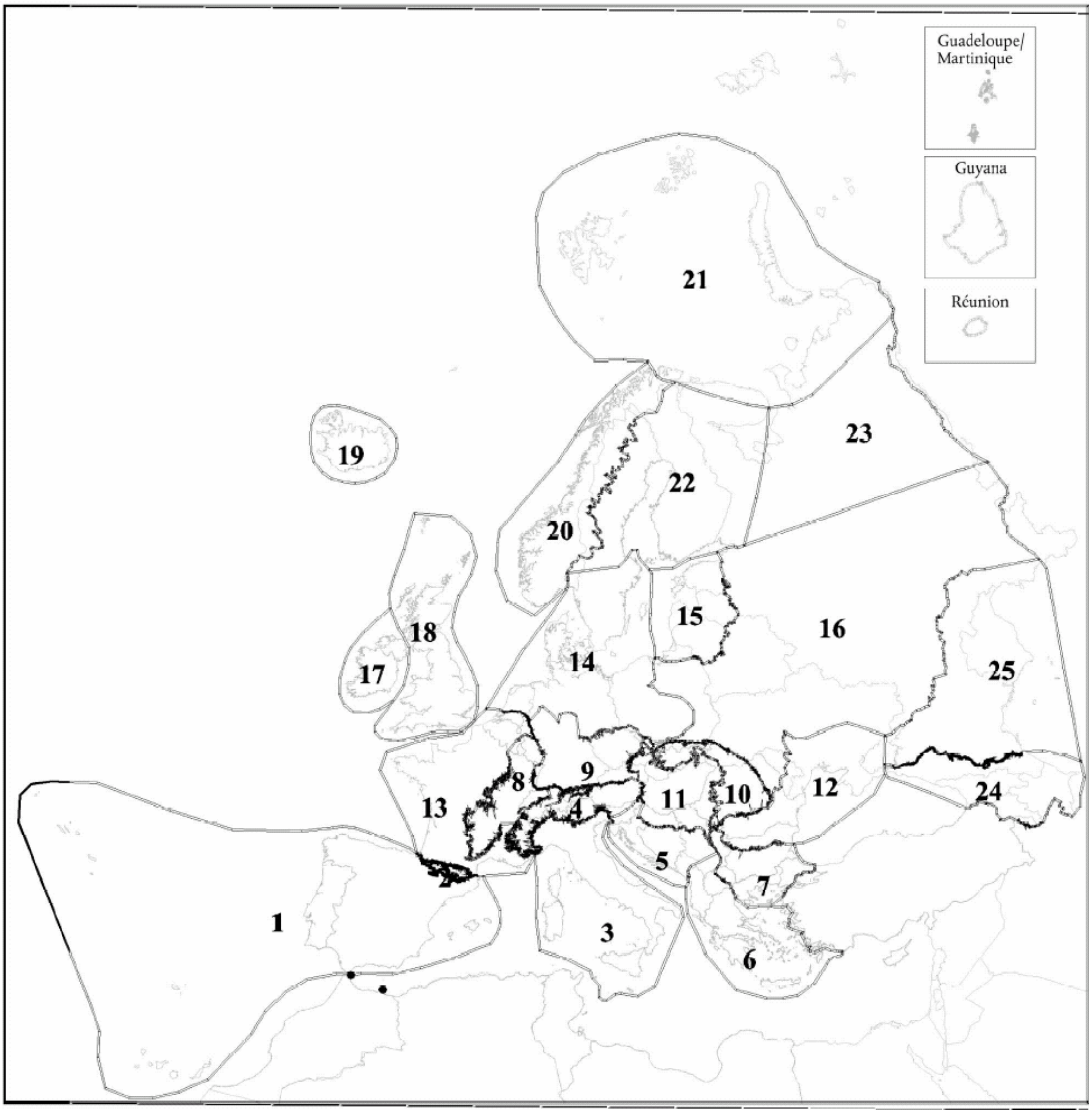
Country/State	Provider	State, County, Administrative District	Provider Type	Web Site
Netherlands	The Netherlands Cadastre and Public Registers Agency		company	http://www.kadaster.com/engels/index.html
Netherlands	Cartography in the Netherlands		university/institution/research	http://www.kartografie.nl/
Netherlands	Expert Center for Taxonomic Identification		university/institution/research	http://www.eti.uva.nl/
Norway	Tromso satellite station		company	http://www.tss.no/
Norway	National Statistic Institute		authority	http://www.ssb.no/english/
Norway	Statens Kartverk		authority	http://www.statkart.no/
Norway	Norwegian Institute of Land Inventory		authority	http://www.nijos.no/English/soil.htm
Poland	National Statistic Institute		authority	http://www.stat.gov.pl/
Poland	Institute of Geodensy and Cartography		authority	http://www.igik.edu.pl/
Poland	Geosystems Polska		company	http://www.geosystems.com.pl/index_en.htm
Portugal	National Hydrology Institute		authority	http://www.hidrografico.pt/
Portugal	Institute for Geology		authority	http://www.igm.pt/Loja/english/default.htm
Portugal	National Statistic Institute		authority	http://www.ine.pt/index_eng.htm
Portugal	The Portuguese National Center for Geographic Information		authority	http://geocid-snig.igeo.pt/Ingles/index.html
Portugal	Swissphoto AG		authority	http://www.swissphoto.ch/html/body_portugal_e.html
Portugal	INAG- Instituto da Água		authority	http://www.inag.pt/
Portugal	Instituto Portugues de Cartografia e Cadastro		authority	http://www.ipcc.pt/portuguese/indexprodu2.html
Portugal	Army Geographic Institute		authority	http://www.igeoe.pt/Geral/ingles/Principal/CENTRO1.html
Portugal	Hydrography Institute		university/institution/research	http://www.hidrografico.pt/
Portugal	Instituto de Hidráulica, Engenharia Rural e Ambiente (IHERA)		authority	http://www.ihera.min-agricultura.pt/
Romania	Geo Strategies, S.A.		company	http://www.geo-strategies.com/
Russia	State Comitee on Statistics		authority	http://www.gks.ru/eng/
Russia	Federal Cadastre Center		authority	http://www.fccland.ru/english.htm
Russia	Russian Ecological Federal Information Agency		authority	http://www.refia.ru/index_e.php?19
Slovakia	Statistics Institute		authority	http://www.statistics.sk/webdata/english/index2_a.htm
Slovakia	National Mapping agency of Slovakia		authority	http://www.gku.sk/index.html

Country/State	Provider	State, County, Administrative District	Provider Type	Web Site
Slovakia	Geodesy Cartography and Cadastre Authority		authority	http://www.geodesy.gov.sk/
Slovenia	Statistics Institute		authority	http://www.sigov.si/
Slovenia	Survey and Mapping Authority of Slovenia		authority	http://193.2.111.28/gu_eng/data/data.asp
Spain	Institut Cartogràfic de Catalunya	Catalunya	authority	http://www.icc.es/cat99/catd/publicacions.html
Spain	Instituto Geologico y Minero de Espana	Catalunya	authority	http://www.igme.es/internet/productos/productosc.htm
Spain	National Statistic Institute		authority	http://www.ine.es/welcoing.htm
Spain	Centre Nacional de Geografia		authority	http://www.cnig.es/
Spain	Instituto Geográfico Nacional		authority	http://www.mfom.es/ign/top_geografico.html
Spain	getmapping.com		company	http://www.getmapping.com/es/
Spain	Spanish Water Information System		university/institution/research	http://hispagua.cedex.es/default_e.htm
Spain	SEISnet		university/institution/research	http://leu.irnase.csic.es/mimam/seisnet.htm
Sweden	National Statistic Institute		authority	http://www.scb.se/eng/index.asp
Sweden	The National Atlas of Sweden		authority	http://www.sna.se/e_index.html
Sweden	Lantmäteriet		authority	http://www.lantmateriet.se/cms/level2index.asp?produktgrupp=105A
Switzerland	Euro Stat data shop Zürich		company	http://www.statistik.zh.ch/europa/
Switzerland	AGIS	Aargau	university/institution/research	http://www.ag.ch/agis/
Switzerland	Umweltdatenkatalog der Schweiz		authority	http://www.ch-cds.ch/d/home.htm
Switzerland	KOGIS		university/institution/research	http://www.kogis.ch/sik-gis/SIK-DI-Bund.htm
Switzerland	Wasser und Energiewirtschaftsamt	Canton Bern	authority	http://www.wea.bve.be.ch/index_d.html
Switzerland	Bundesamt für Umwelt, Landschaftund Wald		authority	http://www.umwelt-schweiz.ch/
Switzerland	Bundesamt für Landestopographie		authority	http://www.swisstopo.ch/
Switzerland	Bundesamt für Wasser und Geologie		authority	http://www.bwg.admin.ch/
Switzerland	Statistik Schweiz		authority	http://www.statistik.admin.ch/
Switzerland	Forum Hydrologie der Schweiz		university/institution/research	http://www.forumhydrologie.ethz.ch/de/
Switzerland	Schweizerische Akademie der Naturwissenschaften		university/institution/research	http://www.sanw.unibe.ch/
Switzerland	Scweizerische Geologische Gesellschaft		university/institution/research	http://www-geol.unine.ch/sgs/welcome_SGS.html
Switzerland	Schweizerische Gesellschaft für Hydrogeologie		university/institution/research	http://www.sgh.ethz.ch/german/index.html

Country/State	Provider	State, County, Administrative District	Provider Type	Web Site
Switzerland	Universität Bern		university/institution/research	http://www.unibe.ch/
Switzerland	ETH Zürich Institute for Cartography		university/institution/research	http://www.karto.ethz.ch/
UK	The Macaulay Institute	Scotland	university/institution/research	http://www.mluri.sari.ac.uk/commercialservices/maps.html
Turkey	State Institute of Statistics		authority	http://www.die.gov.tr/ENGLISH/index.html
UK	Ordance Survey Mapping Northern Ireland	Northern Ireland	authority	http://www.osni.gov.uk/catalog/index.htm
UK	Dotted Eyes		authority	http://dspace.dial.pipex.com/town/terrace/ev90212/aboutus.htm
UK	The Ordnance Survey Great Britain		authority	http://www.ordsvy.gov.uk/
UK	The National Groundwater Level Archive		authority	http://www.nwl.ac.uk/ih/nrfa/groundwater/index.htm
UK	Centre for Ecology & Hydrology		authority	http://www.ceh.ac.uk/data/EIC.htm
UK	National Soil Resources Institute		authority	http://www.silsoe.cranfield.ac.uk/nsri/services/natmap.htm
UK	InfoTech Enterprises Europe		company	http://www.infotech-europe.com/
UK	GDC Digital Mapping Solutions		company	http://www.graphdata.co.uk/products.asp?var_category=data&var_category2=data
UK	The DataStore		company	http://www.data-store.co.uk/search/detail.asp?ProductCode=1970011
UK	CEH Wallingford		company	http://www.nwl.ac.uk/ih/www/research/bhost.html
UK	Simmons mapping		company	http://www.simmonsmap.com/simm.htm
UK	AEROFILMS LTD.		company	http://www.aerofilms.com/company.html
UK	getmapping.com		company	http://www1.getmapping.com/business/intro/flashintro.html
UK	Multimap		company	http://www.multimap.com/static/photoinfo.htm
UK	UK Perspectives		company	http://www.ukperspectives.com/
UK	The DataStore		company	http://www.data-store.co.uk/search/detail.asp?ProductCode=1970009
UK	getmapping.com		company	http://www1.getmapping.com/business/frameset.html

Country/State	Provider	State, County, Administrative District	Provider Type	Web Site
UK	landmap		company	http://www.landmap.ac.uk/
UK	Geomantics, Ltd.		company	http://www.geomantics.com/
UK	Global Insight		company	http://www.europa-tech.com/gi.htm
UK	MapInfo, Ltd.		company	http://www.mapinfo.co.uk/
UK	Nigel Press Associates (NPA Group)		company	http://www.npagroup.co.uk/
USA	National Geodetic Survey		authority	http://www.ngs.noaa.gov/
USA	The National Biological Information Infrastructure		authority	http://www.nbii.gov/
USA	The US Geological Survey		authority	http://www.usgs.gov/
USA	National Science Foundation		authority	http://www.nsf.gov/
USA	Laboratory for Applications of Remote Sensing in Ecology (LARSE)		university/institution/research	http://www.fsl.orst.edu/larse/
USA	Map Mart		company	http://www.mapmart.com/vector.htm
USA	National Imagery and Mapping Agency (NIMA)		authority	http://www.nima.mil/portal/site/nga01/
USA	ESRI		company	http://www.esri.com/
USA	USGS-EROS		authority	http://edc.usgs.gov/
USA	NOAA		authority	http://www.noaa.gov/
USA	GeoCommunity		company	http://www.geocomm.com/
Yugoslavia	Federal Statistic Office		authority	http://www.szs.sv.gov.yu/english.htm
	OMNI map		authority	http://www.omnimap.com/cgi-bin/shop/continue
	Agriculture department of the United Nations		authority	http://www.fao.org/ag/agl/agll/dsmw.HTM
	Agriculture department of the United Nations		geodata_list	http://www.fao.org/ag/guides/resource/data.htm
	The Hydrographic Society		university/institution/research	http://www.hydrographicsociety.org/
	International Office for Water		university/institution/research	http://www.oieau.fr/anglais/index.htm
	Wetlands International		university/institution/research	http://www.wetlands.org/default.htm
	International Organisation for Plant Information		university/institution/research	http://iopi.csu.edu.au/iopi/
	Global Hydrology Resource Center		university/institution/research	http://ghrc.msfc.nasa.gov/
	GeoConcept		company	http://www.geoconcept.com/
	International Soil Reference and Information Centre		university/institution/research	http://www.isric.nl/

ANNEX 3 ECOREGIONS DESCRIBED IN THE WATER FRAMEWORK DIRECTIVE.



Ecoregions for rivers and lakes

1. Iberic-Macaronesian region

2. Pyrenees

3. Italy, Corsica and Malta

4. Alps

5. Dinaric western Balkan

6. Hellenic western Balkan

7. Eastern Balkan

8. Western highlands

9. Central highlands

10. The Carpathians

11. Hungarian lowlands

12. Pontic province

13. Western plains

14. Central plains

15. Baltic province

16. Eastern plains

17. Ireland and Northern Ireland

18. Great Britain

19. Iceland

20. Borealic uplands

21. Tundra

22. Fenno-Scandian shield

23. Taiga

24. The Caucasus

25. Caspic depression

ANNEX 4 TEMPORARY AND EPHEMERAL SURFACE WATERS: EXAMPLE OF THE IBERIAN PENINSULA.

Ten scenarios have been defined in the step 3 of the FOCUS procedure, for which the characteristics of surface water bodies have been identified. The water bodies considered are field ditches, first order streams and ponds, which are permanent water bodies; as a consequence seasonal water bodies and wetlands are not considered in the current risk assessment. However temporary or ephemeral water bodies are often important in agricultural areas: in the north of Europe, drainage ditches often dry out during summer months, and in southern Europe, many unregulated surface waters are ephemeral, only filling during storm events or from seasonal rains. The arid and semiarid regions of the southeast of the Iberian Peninsula are a good illustration of this type of hydrology. Arid and semiarid regions are characterized by a negative water balance and extreme abiotic and biotic variability is probably the main feature of these semiarid ecosystems (Likens 1999). A summary of the actual knowledge about rivers from semiarid regions is included in the paper of Vidal Abarca et al. (2004).

The hydrology of the Mediterranean countries is different from the northern European countries: floods and droughts are alternating events in the Mediterranean climate. The irregularity of the hydrology is determined by the spatial distribution, the annual and seasonal frequency and the occurrence of strong torrents with flows higher than the annual average. The yearly hydrological cycle is alternated by a raining period and a drought period. Therefore the physical, chemical and biological parameters are determined primarily by seasonal hydrological fluctuations. This factor is essential in the dynamics of populations of aquatic organisms and the knowledge of these dynamics is essential for an adequate risk assessment.

There are few important rivers that collect water from a large basin. An important point in Mediterranean countries is the temporal and spatial distribution of the hydrology that leads to the necessity to build dams and reservoirs upstream in the main rivers. Water storage in dams leads in turn to a decrease in flow during the drought season and the little streams that contribute during the raining season can be dry during this period. The river flow is transformed during the drought season in a series of disconnected little ponds that act as reservoirs of aquatic species, which can colonise the river when its flow is restarted. A high percentage of streams can be identified as seasonal streams, with streambeds being dry during the dry season, during which they may be periodically rewetted following intense rainfall and receive run-off from the neighbouring fields. The connection of these streams with permanent streams and water bodies has to be considered in the exposure assessment.

According to Vidal Abarca et al. (2004) there are three key elements that can help to explain the different hydrological behaviour seen in the rivers from semiarid regions. The intermittence or permanence of water in the riverbeds, the presence and type of relationships among surface, subsurface and ground waters and the characteristics of the lithology. As far as hydrology is concerned, three types of rivers can be defined:

- Permanent rivers: the water flow is maintained during the whole yearly water cycle.

- Temporary rivers: the water flow stops during a period in the year

- Ephemeral rivers: the water only flows after heavy precipitation.

In the Iberian peninsula the intermittence increases as a gradient from the more humid north, where the permanent rivers predominate, to the south where temporary and ephemeral rivers prevail (Vidal-Abarce et al., 1992). At a more precise spatial scale, the same river can take part in the three defined types. Therefore, a new concept arises, referred to the intermittent character of some rivers in the semiarid regions. The intermittence also depends on the dominant lithologic substrate of the drainage basin. The substrate permeability is different among the materials, explaining, in many cases, the spatial and temporal distribution of the water.

One of the main consequences of this typical river model is the physical and chemical variability at spatial and temporal scales. The lithology and geomorphology define the physical and chemical characteristics of water, lithology is responsible for the salt content in the surface and ground waters of the different drainage basins. At a more precise scale, the micro-relief, which determines the depth of the water layer, the accumulation of fine sediments or organic matter, the rate of the water stream and the presence of lateral emergences, is related to the observed punctual variations (Gómez el al., 2001).

These types of water bodies contain communities that are quite different from those of permanent water, and that are highly adapted to the changing conditions. They usually include very resilient species with relatively short life-cycles, high mobility and/or desiccation-resistant resting stages (so-called ‘r strategists’) that are able to exploit the high temporal variability in conditions. These types of organism tend to be much more resistant to a variety of disturbances (both physical and chemical) than organisms that are more closely associated with permanent waters (e.g. Townsend et al., 1997).

In general, bio1ogical communities predominantly respond to the spatial and temporal variability in water and the salinity of the arid and semiarid rivers. The riverside vegetation is distributed from up-waters to down-water, showing the water discontinuance of these rivers. They also respond to the variances in salinity and the depth of the water layer. Thus, the quality and quantity of the water are the main factors for the distribution and composition of the riverside vegetation. The adaptations of algae and macrophytes are related to the tolerance of salinity and drought periods.

Many aspects of the dynamics of these rivers remain unknown. However, the water conditions imply certain particularities in the function. Thus, in primary production terms, dryness increases autotrophy (Gasith and Resh, 1999). Autotrophic metabolism is favoured by reduction of the volume of water; these rivers are producer-accumulators of pulse-like exported organic carbon during periods of rising water levels and freshets (Vidal-Abarca et al., 2004).

Among the aquatic invertebrates, insects are the best group adapted to these stressful factors, e.g. Diptera and Coleoptera, which are well represented. Among them, there are a number of species of ecological and/or biogeographic interest (Vidal-Abarca et al., 2004). There are some reports on the typical species that live in the water bodies from semiarid regions (Moreno et al 1997; 2001; Oliva-Paterna et al., 2003). In the following list some species typical from rivers of semiarid regions are mentioned (Vidal-Abarca et al., 2004):

Trophic group	Species
Primary producers	*Ruppia drepanensis; Vaucheria sescuplicaria, Mougeotia faurelii*
Invertebrates	*Ochthebius ontesis, O. Glaber, O. Delgadoi, Nebrioporus baeticus, agabus ramblae, Eretes Sticticus, Cybister lateralimarginalis, Sigara scripta, Coenagrion mercuriale*
Predators	*Aphanius iberus, Barnus sclateri*

In this context, wetlands should receive particular attention. Wetlands may receive water from several sources: overland flow, precipitation, and groundwater discharges, and the source of water affects both water chemistry and hydrologic period (flooding depth, frequency, duration, and seasonality). Wetland ecosystems, by definition, depend on water to maintain their ecological functions. The hydrological cycle renews the flow and quantity of water in rivers, aquifers, lakes and all other freshwater ecosystems. These are complex ecosystems, the boundaries of which are often in a state of flux. Wetlands are therefore easily affected by external events. Nutrient and sediment loads, for example, are frequently moved from one site to another, and from one habitat to another. While the fluid nature of such exchanges guarantees a continued renewal of energy, it also represents a major potential hazard since many harmful agents (pesticides, fertilisers or other chemicals) can also be easily and rapidly transported to other areas where they might have an adverse impact on the environment. The high productivity of wetlands also provides support for large numbers of birds, many of which depend on a network of wetland sites during long seasonal migrations, or as breeding or overwintering grounds.

One of the best examples of wetlands in Spain is the Doñana National Park (SW Spain), during flooding, large areas are covered with wetlands that desiccate during drought, totally about 24,000 ha of marshes and 300 seasonal ponds ranging from 0.01 to 40 ha size. These seasonal ponds are fed by a combination of ground water and rainfall, and the relative amount of each water input varies during wet and dry periods (Sacks et al., 1992). They are diverse and productive environments, although highly variable in time. Hydrological variations can be so large that these wetlands rarely resemble themselves from one year to the next due to major changes in water level, flooded area, water colour, macrophyte development and surrounding vegetation (Serrano and La Toja, 1995). As a result, the annual variability of both phytoplankton and zooplankton can be very large.

Wetlands are reservoirs of some fish species that can colonise rivers. Some examples of fish species are given below:

Fish species	Description
Cyprinus carpio	A very adaptable species that may be found in all types of habitats apart from streams, tolerant to contamination and lack of oxygen
Carassius auratus	A species being very adaptable and resistant to contamination, living in backwater with vegetation
Barbus bogagei	An autochthonous species of the Iberian Peninsula that may be found in rivers and wetlands
Barbus comiza	A typical species of big rivers, deep waters, dams and lakes, endemic of Guadalquivir, guadiana and Tajo rivers
Chondrostoma polypeis	A very sensitive species used as a biological indicator, being typical of the medium course of the rivers
Leuciscus pyrenaicus	An endemic species of the centre and south of the Iberian Peninsula that may be found in wetlands
Rutilus arcasii	An endemic species of some rivers, the populations in the wetlands are in the process of disappearance
Tropidophoxinellus alburnoides	An endemic of the Iberian Peninsula, being very adaptable and able to colonise seasonal streams. The populations in the wetlands are in the process of disappearance
Ictalurus melas	Living in backwater, this species is very adaptable and resistant to the lack of oxygen, to contamination and to the desiccation, and can survey buried under the mud
Lepomis gibbosus	A species typical in wetlands
Gambusia affinis	A species that colonises all types of wetland and rivers

Wetlands and temporary (seasonal) surface waters in many cases do not meet the recognition of permanent water bodies as for example for protection status. Nevertheless, seasonal water bodies play a role as surface water reservoirs or derivation drains each of which may gain importance for further irrigation purposes.

Little knowledge is available concerning whether or not communities living in such ecosystem are more or less sensitive than communities from permanent water bodies. Nevertheless, it appears clear that a typical scenario dedicated to temporary wetlands and streams should be developed at least for environmental, i.e. PEC calculation, purposes.

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ANNEX 5 ECOLOGICAL TRAITS FOR CRUSTACEAN SPECIES (FROM TACHET ET AL., 2000)

Table 1: Ecological traits described with affinity scores (from 0 to 3 or 0 to 5) for each modality. For details on modalities see Table 2. For details on affinity scores see Table 3.

Trait n°	1						2		3			4			5			6			7			8							
modality	2	3	4	5	6	7	1	2	1	2	3	1	2	4	1	3	4	1	2	3	1	4	5	2	3	4	5	6	7	8	9
Anostraca		2	3	1			3				3	2	3	3	1					3	3			3	3						
Argulus		3	1				3				3	2	3	3			3	3	1				3								3
Asellus aquaticus			3				3	1		1	3	3	3	3	3			3				2	2	1	3	1	1				
Astacus						3		3		3		3	3	3	3		2		2				3		1		1			2	2
Atyaephyra desmarestii				3			1	3		3		3	3	3	3			1	1				3	1	3				1		
Austropotamobius						3		3	3			3	3	2	3		3		2		2	2			3		2	2		2	
Conchostraca			3				3				3	3	3	3	3					3	3			3	3						
Corophium		3					3				3	3	3	3	3			2	2				3	2	2	1					
Crangonyx	2	2						3			3	3	3	3	3			1					3		4	3					
Echinogammarus				3				3			3	3	3	3	3			3	1				3	1	5	2	1	2	2	1	

Eriocheir sinensis					3	2		3		3		3	3	2		3			3				3				3			3	1
Gammarus			2	2			1	3			3	3	3	3	3			3	2				3	1	5	2	1	2	2	1	
Lepidurus apus						3	3				3	2	3	3	1		3			3	3			3	3	1			1		
Niphargidae				3				3	3			3	3	3	3			3					3	5	1				1	1	
Orconnectes limosus					1	3		3		3		3	3	3	3		3		2		3				3					3	
Pacifastacus leniusculus						3		3		3		3	3	3	3		3		2				3				3		1	2	
Potamon ibericum					3			3		3		3	3	2	3				1				3		2		1	2		3	
Proasellus		3					1	3		1	2	3	3	3	3			3				2	2	1	3	1	1				
Procambarus clarkii						3		3		3		3	3	2	3		3		1				3				3				2
Triops cancriformis					2	3	3				3	2	3	3	1		3			3	3			3	3	1			1		

Trait n°	9							10		11			12						13			14					15		16				
Modality	2	3	4	5	6	7	8	1	2	1	2	3	1	2	3	4	5	6	1	2	3	1	2	3	4	5	1	2	1	2	3	4	5
Anostraca	2			3				1	3		2	3									3	1	1	1	1	1	3	2		2	2	2	2
Argulus					2		3	1	3		2	2							1	1	1						3			3	1	3	
Asellus aquaticus		3							3			3	1	1	1	2	3	3	1	3	2			2	3	1	3	2		3	3	2	

Astacus	1	1				2		3	2	2	2			1	1	1	1	3	1			1	2	1		3	1	2	1	2	2	1
Atyaephyra desmarestii		3	1					3		1	3					1	3		2	3		1	3	1		3	1		1	1	3	1
Austropotamobius		3						3	3	1	1							3			1	1				3		3	3	3	3	3
Conchostraca	3			3			1	3		2	2									3	1	1	1	1	1	3				1	2	2
Corophium	2	3						3		1							3		2	1		1	3	1		3	3				3	
Crangonyx			3					3		1							3		2	1						3	1			3	3	
Echinogammarus		3						3		1				1	1	2	2	2	2			2	2			3	1	2	2	2	2	2
Eriocheir sinensis		3				3		3	1		2									2						2	3			1	3	
Gammarus		3	1					3	1		3	1	1	2	2	2	3	3	3		1	2	3	2		3	1	2	2	2	2	2
Lepidurus apus	3					1	1	3		2	3					1	3			3	1	1	1	1	1	3					3	3
Niphargidae		3					1	3	3		1			1	1	1	3	2			3	1	1			3		3	3	3	3	3
Orconnectes limosus		2	2			2		3			3						3	2	3				3	2		3		1	2	3	3	2
Pacifastacus leniusculus		3				2		3			3							3								3			3		2	1
Potamon ibericum		3				3		3		3	2							3								3	2					3
Proasellus		3						3			3	1	1	2	2	3	3	1	3	1			1	3		3	1			2	2	2
Procambarus clarkii		3				3		3			3								1	3				3		3					2	1
Triops cancriformis	3					1	1	3		2	3									3	1	1	1	1	1	3				3	3	

Trait n°	17			18								19							20									21					22			
Modality	1	2	3	1	2	3	4	5	6	7	8	1	2	3	4	5	6	7	1	2	3	4	5	6	7	8	9	3	4	5	6	7	1	2	3	4
Anostraca	3	3	2								3			2	2	3						2				2	3	3					3			
Argulus	3							1	1		3		1	3			3		1	1	1	1	1	1	1	1	1	1				3	3	2		
Asellus aquaticus	3			2	2	2	2	2	2	2	2	3	5	3			2		1	1	1		5	3	2	3	1		3		2		3	3		
Astacus	3				1	2	2	1			1		2	2			1		2		1						1	1	4	2			2	2	1	
Atyaephyra desmarestii	3							2	3		2	1	3	1	3	1	2		1				3					5					3	2		
Austropotamobius	3				3	3					1		2	2			1		3	1					1			1	4	1				1	3	2
Conchostraca	3										3			1	1	2	1										3		3	2			3			
Corophium	3							2	2	2		2	2						3			3	3		3	1		1	3	1			3	3		
Crangonyx	2							1	1	1		3	2	3			3					1	3			3		2	3		1		3	3		
Echinogammarus	3							1	3	2		3	4						5	3	1	1	5	2	4	4		2	3					2		
Eriocheir sinensis	3							2	3	3			3	2					2		1		2					1	3	3			3	2		
Gammarus	3	3		2	2	2	2	2	2	2	1	3	4	1		1	2		5	3	2	1	3	2	4	4		2	3		1		1	2	3	1
Lepidurus apus	3										3						2					2				2	3	2	3	1			3			
Niphargidae	3	1									3		1		1			5	1	1	3	5						1	3	1	3		3	1		

Orconnectes limosus	3				1	2	2	2	2		1		2	2			1		2			2	3					1	4				2	3	1	
Pacifastacus leniusculus	3	1			1	1	1				1		3	2			1		1				3					1	4	1				2	2	
Potamon ibericum	3										3		3						3				3			2			2	3			3	2		
Proasellus	3			3	3	1	1	1	1	1	1	3	3	2		2				3			4	3					3		2		3	2	1	
Procambarus clarkii	3										3			1	3	1							3						1	3			3			
Triops cancriformis	3										3					3						2				2	3	2	3	1			3			

Table 2: Description of ecological traits through modalities.

Traits 1 to 22	Code of the modality	Nature of the modality
1 size (max)	1	< 2.5
	2	2.5-5
	3	5-10
	4	10-20
	5	20-40
	6	40-80
	7	>80
2 life cycle	1	≤ 1 year
	2	> 1 year
3 number of generation/year	1	<1
	2	1
	3	>1
4 aquatic stage	1	Egg
	2	Larvae
	3	Imago
	4	Adult
5 reproduction	1	Ovoviviparity, young care
	2	Isolated, free eggs
	3	Isolated, fixed eggs
	4	Egg laying, free
	5	Egg laying, fixed
	6	Endiphytic egg laying
	7	Terrestrial egg laying
	8	Asexual reproduction
	9	Parthenogenesis
6 dispersal	1	Aquatic, passive
	2	Aquatic, active
	3	Aerial, passive
	4	Aerial, active
7 resistance forms	1	Eggs, plumule, statoblasts, shell
	2	Cocoons
	3	Box against dessication
	4	Diapause or quiescence
	5	no

8 food	1	Fine sediment + microorganisms
	2	Debris < 1 mm
	3	Vegetal debris > 1 mm
	4	Microphytes (alive)
	5	Macrophytes (alive)
	6	Animals (dead)
	7	Microinvertebrates (alive)
	8	Macroinvertebrates (alive)
	9	Vertebrates
9 feeding mode	1	Absorption through teguments
	2	Fine sediments
	3	Shredders
	4	Scrappers, grazers
	5	Filterers
	6	Piercers (algivorous or sucking predators)
	7	Predator
	8	Parasite
10 respiration	1	Tegument
	2	Gill
	3	‘plastron’
	4	spiracles
	5	hydrostatic vesicles
11 temperature	1	Stenotherm psychrophyle (< 15℃)
	2	Stenotherm thermophyle (< 15℃
	3	eurytherm
12 pH	1	< 4
	2	4-4.5
	3	4.5-5
	4	5-5.5
	5	5.5-6
	6	> 6
13 thophic degree	1	Oligotrophic
	2	Mesotrophic
	3	Eutrophic
14 saprobic value	1	Xenosaprobic
	2	Oligosaprobic
	3	Beta mesosaprobic
	4	Alpha mesosaprobic
	5	Polysaprobic
15 salinity	1	Soft water
	2	Brackish water
16 biogeographic areas (limnofauna europaea)	1	2 : Pyrénées
	2	4 : Alpes and Jura
	3	8 : Massif central and Vosges
	4	13a : low land (oceanic)
	5	13b : low land (Mediterranean)

17 altitude	1	Plain + hill (> 1000 m)
	2	Mountain (1000-2000 m)
	3	Alpine (> 2000 m)
18 longitudinal distribution	1	Crenon
	2	Epirhithron
	3	Metarhithron
	4	Hyporhithron
	5	Epipotamon
	6	Metapotamon
	7	Estuary
	8	Out of fluvial system
19 transversal distribution (from open channel)	1	River
	2	Bank, secondary channels
	3	Ponds, pools, meanders
	4	Marsh, peat bog
	5	Temporary waters
	6	Lakes
	7	Ground waters
20 microhabitats (preference)	1	Flagstone, block, stone, pebblestone
	2	Gravel
	3	Sand
	4	Silt
	5	Macrophytes, filamentous algae
	6	Microphytes
	7	Branch, root
	8	Litter
	9	mud
21 locomotion	1	Fly
	2	Surface swimmer
	3	Plankton, necton
	4	Crawler
	5	Benthic, epibenthic
	6	endobenthic
	7	temporary fixed
	8	permanently fixed
22 current	1	No
	2	Slow (< 25 cm/s)
	3	Medium (25-50 cm/s)
	4	Fast (> 50 cm/s)

Table 3: Method for affinity score calculation.

Code of modality	Nature of modality	Affinity score	%
1	Crenon	0	0
2	Epirhithron	1	11
3	Metarhitron	1	11
4	Hyporhithron	3	33
5	Epipotamon	3	33
6	Metapotamon	1	11
7	Estuary	0	0
8	Out of fluvial system	0	0
	Total	9	100

Reference

Tachet H., Richoux P., Bournaud M. and Usseglio-Polatera P., 2000. CNRS ed.: Invertébrés d'eau douce: systématique, biologie, écologie. Paris, 587 pp.

ANNEX 6 COMPARISON BETWEEN BIOLOGICAL INDICES FOR INVERTEBRATE TAXA (IBGN) AND TAXA SENSITIVITY RANKING TO ORGANIC SUBSTANCES

In their study, Statzner *et al* (2001) observed some correspondence between the invertebrate abundance data for the 10 most natural regions in France and those found in other natural European regions. The polluosensitivity index based on invertebrate relative abundance data (described above) was compared to available sensitivity ranking to toxicants in order to evaluate the comparability of the two ranking methods Comparison of the IBGN polluosensitivity index and the database ranking of Wogram and Liess (2001) provided a poor correlation. This was not surprising since the IBGN classification is based on non-specific compounds while the ranking of Wogram and Liess is based on sensitivity to active compounds. Furthermore, the level of identification was not similar in the two databases thus limiting analytical sensitivities. Finally, many factors may influence the survival of a population in aquatic ecosystems, which are not taken into account in laboratory tests for species sensitivity ranking databases.

Other attempts have been made to relate acute toxicity test results with abundance and diversity of benthic fauna (Long *et al.,* 2001). The authors found a good correlation between percentage survival of amphipods (*Rhepoxynius abronius* or *Ampelisca abdita*) in sediment toxicity tests in the laboratory and both the number of species and the total abundance of benthic invertebrates in marine water samples.

Satisfactory correlations were observed for some species between laboratory tests results and field impact measurements in sediment contaminated with PAH and metals (Ferraro and Cole, 2002). As an example, relationships between survival of *Rhepoxynius abronius* and *Leptocheirus plumulosus* exposed to field sampled sediment in the laboratory, and four endpoints (number of species, numerical abundance, Swarz dominance index and Brillouin index) measured in the field showed correlation coefficient ranging from 0.31 to 0.51. Correlation with biomass in the field was not demonstrated. Good agreement was also observed between acute toxicity tests of field sampled sediment (contaminated with PCB/PAH and metals) with the amphipod *Leptocheirus plumulosus* in the laboratory (% survival measured) and the effects on population (density) measured at the tests sites (McGee *et al.*, 1999).

In situ measurements of survival rates in caged adults *Gammarus pulex* and larval *Limnephilus lunatus*, compared to population monitoring showed that good agreement could be achieved for some species between caged and stream data (*L. lunatus*). Overestimation of acute effects in caged individuals could nevertheless be observed because of difference in behaviour between caged and free populations of *G. pulex* (Schulz and Liess, 1999).

Biological indices for macroinvertebrates used to assess impact of copper and zinc to a stream mesocosm appeared to be inappropriate to distinguish metal polluted and control ecosystems (Hickey and Golding, 2002). In another study, biological indices (Ephemeroptera-Plecoptera-Trichoptera - EPT- richness) appeared to be more sensitive than water sediment test with *D. magna*, *C. dubia* and a water column test with the Asian clam to distinguish between contaminated (acid mine drainage) and non-contaminated ecosystems (Schmidt *et al.*, 2002).

Biological traits themselves (e.g. lifespan, dispersal, or aquatic stage) seemed to be more adapted to distinguish between a chemical, an organic and a physical pollution (Charvet et al, 2001). The authors identified relationships between biological traits representative of e.g. resistance or resilience of communities and environmental characteristics of their habitat. Traits were then used to discriminate between different types of pollution. Biological traits proved to be more stable in space (independent of the size of a river) and time than were taxonomy based indicators.

As far as phytoplankton is concerned, differences in sensitivity of diatoms to trophic changes (basis of diatom index) and metal stress in a stream may be very different (Ivorra *et al*., 2002). In their study, diatom community structure was found to respond to both nutrients and metal exposure. Under both field and laboratory metal polluted conditions, indices did not react according to predictions based on trophic considerations. Dorigo *et al*. (2004) reached similar conclusions when comparing Diatom Indices (BDI), polluosensitivity indices (PSI) and laboratory tests for algae exposed to atrazine. Combined input of toxicants and nutrients may be related to increase in intraspecific variation to toxicants in ecosystems as for example among phytoplankton (Selck *et al.*, 2002). As for other trophic groups, an important variability among species sensitivity is recognised (Blanck *et al.*, 1984; Wänberg and Blanck, 1988), even in the same genus (Tuckey *et al.*, 2002).

In general, there are some doubts on the relevance of polluosensitivity indices to distinguish between chemically (i.e. toxicologically) polluted and unpolluted areas as tolerance to pollution (in the ecological sense of being found in polluted areas) does not imply necessarily a tolerance to toxicants (Nalepa and Landrum, 1988, Charvet et al., 2001).

The work of Stuijfzand (1999) compared the sensitivity to toxicants of two species (*Chironomus riparius* and *Dreissena polymorpha*) which are quite different in tolerance to organic pollution. From this study, the hypertrophic midge *C. riparius* was not a suitable species to assess adverse effects of pollution on macrofauna species especially with the presence of toxicants coincides with organic enrichment. Particulate matter reduced the effects of inorganic pollutants by reducing bioavailability but also by stimulating growth. The success of *C. riparius* to develop was not explained by tolerance to toxicants *per se* but rather by its ability to take advantage of organic enrichment associated with the presence of toxicants.

Nevertheless, such ranking studies allow to see some trends in global sensitivity of species. As an example, Cladocera belong, among invertebrates, the most sensitive taxa to chemical pollution (Wogram & Liess (2001).

It may happen that these trends are not confirmed under experimental conditions. For example, mesocosm studies performed with insecticides where *Daphnia magna* was not the most sensitive species even when the Cladocera were the most sensitive taxa to the tested insecticides in laboratory tests (van den Brink, 2002).

Nevertheless, this does not preclude of the usefulness of species sensitivity ranking in risk assessment. In the example above (van den Brink, 2002), the 5th percentile of acute SSD for arthropods coupled with an application factor of 10 was protective compared to the NOEC for the most sensitive taxa in the microcosm experiments. Other authors that have tried to compare both strategies for risk assessment had similar conclusions. Selck *et al*. (2002) found ratios between the lowest NOEC from enclosure experiments with tributyltin and linear alkyl benzene sulfonates (LAS) and PNEC calculated on the basis of laboratory effect data ranging from 1.5 to 6.7 for TBT and from 10 to 1733 for LAS. Thus the lowest effect concentration measured in the field was higher than extrapolated (application factor-AF and SSD) PNEC values in all cases. It appeared that few acute toxicity data and large AF did not give the lowest PNEC, since for TBT SSD provided the most conservative PNEC. This study led to the conclusions that the application factor provided an adequate estimate of effects levels in both cases where few or more data exist and that SSD models were the most conservative when a larger data set exists. Forbes *et al.* (2001) compared output of simulations with PNEC derived by an application factor and a log-normal distribution based model of NOEC for juveniles and showed that current extrapolation approaches may be over protective but conditions also exist where this is not the case. In general, information for plankton and macrophytes and for fish as well is more limited so that it remains difficult to draw general inferences.

Even though many parameters were measured in the Meuse, these were not predictive for the negative or positive effects of the river water on Macrofauna. This confirms the results of Hendriks *et al.*, 1994 in Stuijfzand, 1999) that more than 89% of the observed effects on bacteria could not be attributed to identified compounds.An attempt to compare the ranks of a same taxa between the IBGN ranking and the species sensitivity ranking available in Wogram and Liess (2001) has been performed and is presented below.

Data were made comparable between the ranks by affecting the same sensitivity rank to different plecoptera taxa as identified in the IBGN. These data are presented in the following table:

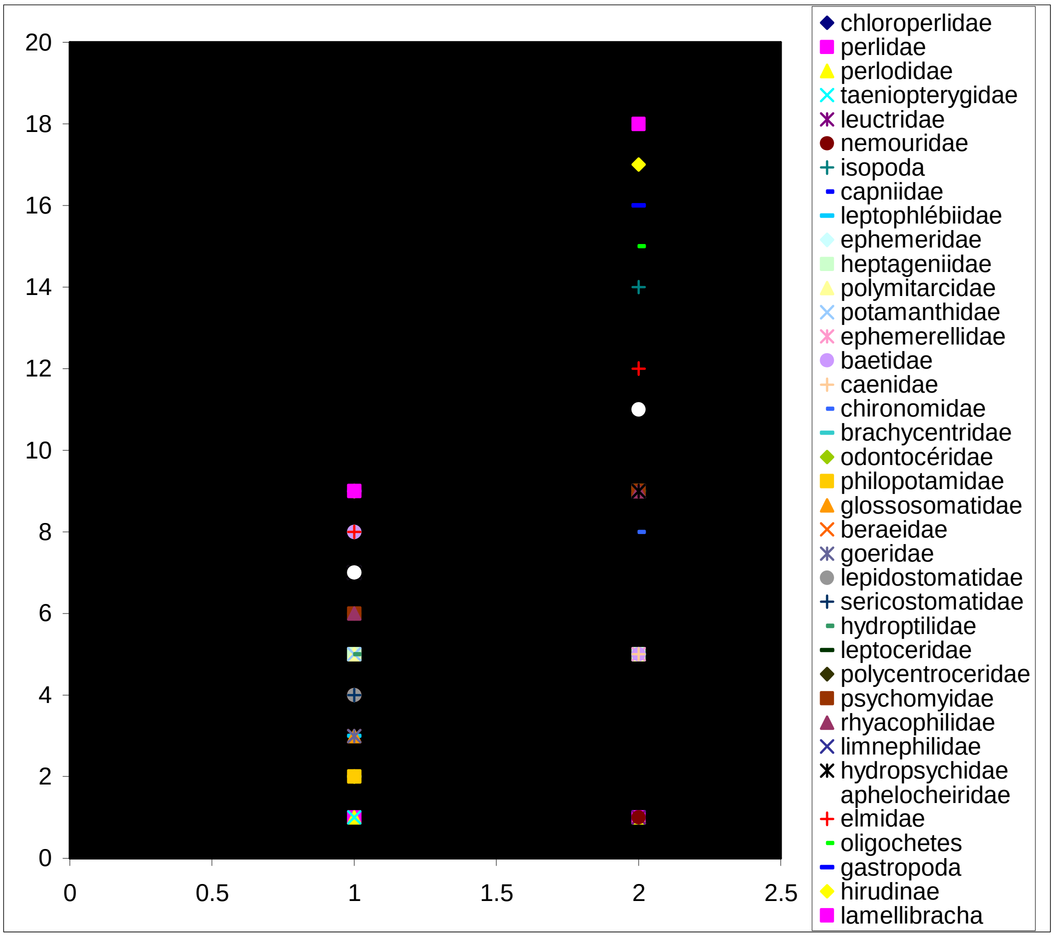
Taxa	IBGN	Sensitivity ranking to organic chemicals
chloroperlidae	1	1
perlidae	1	1
perlodidae	1	1
taeniopterygidae	1	1
leuctridae	3	1
nemouridae	4	1
isopoda	9	14
capniidae	2	5
leptophleiidae	3	5
ephemeridae	4	5
heptageniidae	5	5
polymitarcidae	5	5
potamanthidae	5	5
ephemerellidae	7	5
baetidae	8	5
caenidae	8	5
chironomidae	9	8
brachycentridae	2	9
odontocéridae	2	9
philopotamidae	2	9
glossosomatidae	3	9
beraeidae	3	9
goeridae	3	9
lepidostomatidae	4	9
sericostomatidae	4	9
hydroptilidae	5	9
leptoceridae	6	9
polycentroceridae	6	9
psychomyidae	6	9
rhyacophilidae	6	9
limnephilidae	7	9
hydropsychidae	7	9
aphelocheiridae	7	11
elmidae	8	12
oligochetes	9	15
gastropoda	9	16
hirudinae	9	17
lamellibracha	9	18

Only paired results have been retained. These two ranges have then been compared according to the following statistical procedure:

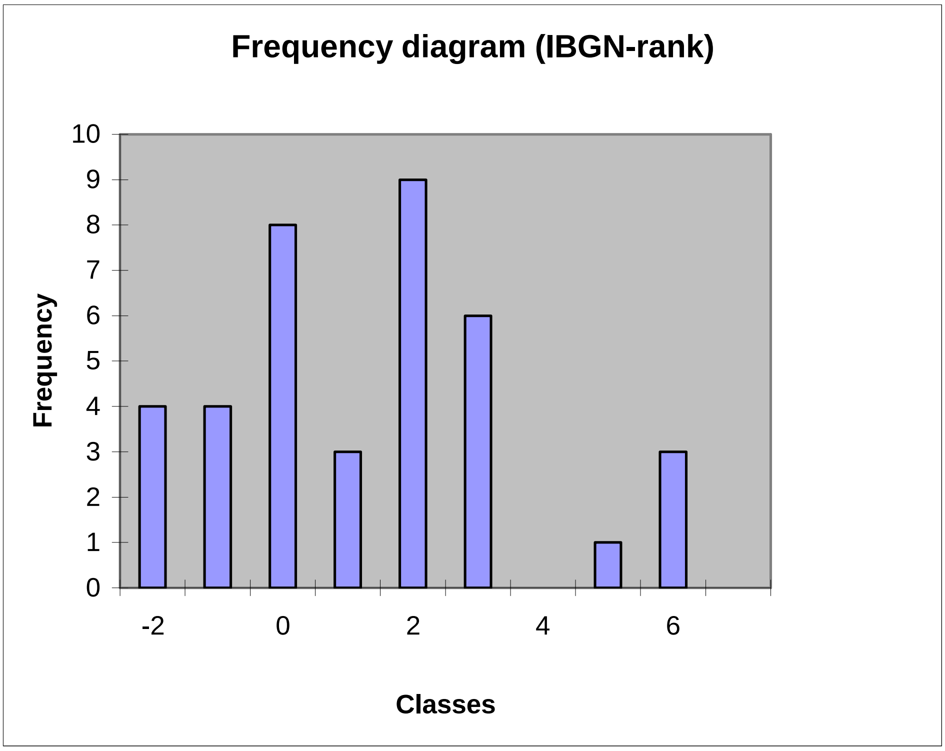
Sample 1: IBGN, 38 data between 1.0 and 9.0

Sample 2: Rank orga, 38 data between 1.0 and 18.0.

The following figure represents the two classifications for all taxa:



Data were then transformed in order to show similar ranges of rank values in order to be compared using a Wilcoxon-Mann Whitney test. Indeed, most analysis of variance statistics may not be suitable to deal with the data. Non-parametric tests are not conditioned by these statistical requirement. The following figure illustrates the distribution of the calculated differences between ranking values:



Conclusions

The Wilcoxon Mann Whitney statistics were used to test the null hypothesis that taxa showed similar ranks in the two classification systems. The z (absolute) value was 1.816, being above 1.64 so that the null hypothesis is rejected (5% threshold) but is lower than 2.33, which is the 1% threshold to accept the null hypothesis. It is important to note that the data set used in these statistics integrates too much transformed data to be easily interpreted. The test was rather performed in order to provide a first comparison between two complete series of data.