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Foreword

The present literature review was prepared within the context of the work package WP1 (‘Integrated knowledge reviews’) of the FOOTPRINT project.

The preferred reference to the present document is as follows:
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Summary

In this document, the current knowledge on mitigation strategies to reduce pesticide inputs into surface water and groundwater, and their effectiveness when applied in practice is reviewed. Apart from their effectiveness in reducing pesticide inputs into ground- and surface water, the mitigation measures identified in the literature are evaluated with respect to their practicability. Those measures considered both effective and feasible are recommended for implementing at the farm and catchment scale. Finally, recommendations for modelling are provided using the identified reduction efficiencies.

Roughly 180 publications directly dealing with or being somehow related to mitigation of pesticide inputs into water bodies were examined. There are considerably more mitigation measures (and literature on mitigation) available for the pathways runoff/erosion and spray drift than for drainage and leaching.

The effectiveness of grassed buffer strips located at the lower edges of fields has been demonstrated. However, the effectiveness is very variable, and the variability cannot be explained by strip width alone. Riparian buffer strips are most probably much less effective than edge-of-field buffer strips in reducing pesticide runoff and erosion inputs into surface waters. Constructed wetlands are promising tools for mitigating pesticide inputs via runoff/erosion and drift into surface waters, but their effectiveness still has to be demonstrated for weakly and moderately sorbing compounds. Subsurface drains are an effective mitigation measure for pesticide runoff losses from slowly permeable soils with frequent waterlogging.

For the pathways drainage and leaching, the only feasible mitigation measures are application rate reduction, product substitution and shift of the application date.

There are many possible effective measures of spray drift reduction. While sufficient knowledge exists for suggesting default values for the efficiency of single drift mitigation measures, little information exists on the effect of the drift reduction efficiency of combinations of measures. More research on possible interactions between different drift mitigation measures and the resulting overall drift reduction efficiency is therefore indicated.

Point-source inputs can be mitigated against by increasing awareness of the farmers with regard to pesticide handling and application, and encouraging them to implement loss-reducing measures of "best management practice". In catchments dominated by diffuse inputs at least in some years, mitigation of point-source inputs alone may not be sufficient to reduce pesticide loads/concentrations in water bodies to an acceptable level.

Results of the present review work will be integrated in the FOOT tools used at the local (FOOT-FS) and the catchment/regional scale (FOOT-CRS) to recommend mitigation measures to reduce pesticide contamination of water resources.

1 INTRODUCTION

The contamination of water bodies with agricultural pesticides can pose a significant threat to aquatic ecosystems and drinking water resources (e.g. Dabrowski et al., 2002). However, the risk for the aquatic community or for human health can often be substantially reduced by appropriate measures (Kreuger and Nilsson, 2001). Mitigation of pesticide inputs into water bodies includes both reduction of diffuse-source (runoff and erosion, tile drainage, spray drift, leaching to groundwater) and of point-source inputs (mainly farmyard runoff), which in some regions of Europe (e.g. Western Germany, Sweden) have been shown to make a highly significant contribution to the observed pesticide loads in rivers (Jaeken and Debaer, 2005).

In this paper, the current knowledge on mitigation strategies to reduce pesticide inputs into surface water and groundwater, and their effectiveness when applied in practice is reviewed. The term “mitigation” is used here in a broad sense synonymously to “risk reduction”, which comprises all measures that lead to a lower risk, i.e. reduce exposure and/or effects. This includes also switching to another pesticide with more favourable physical/chemical or ecotoxicological properties.

A number of extensive reviews on a range of mitigation measures and their effectiveness already exist (e.g. Norris, 1993; Dosskey, 2001, Ucar and Hall, 2001; FOCUS, 2004b; Schulz, 2004; Lacas et al., 2005; Krutz et al., 2005). However, a compilation of the efficiencies of the mitigation measures available for the different pesticide input pathways is lacking so far. Furthermore, apart from their effectiveness in reducing pesticide inputs into ground- and surface water, there is a need for an evaluation of the practicability of the various mitigation measures and advice whether they should be considered for implementation in practice or not. Furthermore, for most mitigation measures there are no recommendations available how to account for their effect in modelling for risk assessment and risk management purposes. This present review therefore aims at

- estimating the efficiencies of the various mitigation measures at the farm scale for different combinations of pesticide properties, soil and climate,
- assessing the effects at the regional/catchment scale due to the implementation of a given mitigation measure,
- assessing the effects of realistic combinations of mitigation measures at regional/catchment scale,
- evaluating the mitigation strategies identified in the literature with respect to their practicability and cost-effectiveness, and recommending those considered both effective and feasible for implementation at the farm and catchment scale,
- providing recommendations for modelling using the identified reduction efficiencies.

2 INPUT PATHWAYS OF PESTICIDES INTO GROUND- AND SURFACE WATER AND POSSIBLE MITIGATION MEASURES

Pesticides can enter water bodies via diffuse or via point sources (Carter, 2000). Diffuse and point sources are not unequivocally defined in the literature (cf. Jaeken and Debaer, 2005), and often a clear distinction between the two is not possible. In accordance with Carter (2000), we define diffuse-source pesticide inputs into water bodies as inputs resulting from agricultural application on the field. In contrast, point-source inputs derive from a localized situation and enter a water body at a specific or restricted number of locations. According to this definition, diffuse input paths for pesticides into surface waters are tile drain outflow, baseflow seepage, surface and subsurface runoff and soil erosion from treated fields, spray drift at application, and deposition after volatilization. Diffuse pesticide input paths into groundwater are leaching through the soil and unsaturated zone, and infiltration through river banks and beds. Point sources are mainly farmyard runoff (either directly into streams or into the sewer system), sewage plants, sewer overflows, and accidental spills. There are also point sources of pesticides from non-agricultural use, e.g. from application on roads, railways or urban sealed surfaces such as parking lots. However, only agricultural sources of pesticide contamination will be considered in the following. The most important input pathways of agricultural pesticides into water bodies and possible mitigation measures for these pathways are briefly explained in the following.

2.1 Surface runoff and erosion

Surface runoff can in principle occur on almost every arable field, even in nearly flat terrain (Leonard, 1988; Wauchope, 1978); yet its frequency of occurrence will depend on the climate. There are essentially two types of surface runoff: *Infiltration excess* or “*Hortonian*” *runoff* is generated when both infiltration capacity and surface storage capacity of the soil are exceeded by the incoming precipitation. Infiltration capacity decreases with increasing silt and clay contents (lower saturated conductivity of the soil matrix), but increases with increasing soil structure and the presence of macropores at the surface. Thus, clay soils with abundant macropores (e.g. shrinking cracks and earthworm channels) at the soil surface can exhibit as high infiltration capacities as coarse-textured soils (Jarvis and Messing, 1995). As silty soils (e.g. loess soils) are very prone to structural degradation by compaction (wheel tracks) or raindrop impact, they are especially vulnerable to Hortonian runoff. In contrast to infiltration excess runoff, *saturation excess runoff* occurs when the soil is saturated, in which case any rainfall onto the soils immediately runs off (Garen and Moore, 2005). Saturated areas typically form at the base of hillslopes, where soil moisture is high due to downslope movement of subsurface water (“interflow”). Surface runoff usually starts as laminar sheet flow and after a certain travel length channelizes to concentrated, turbulent flow (Hillel, 1980).

Soil erosion by water consists of two processes: i) the detachment of soil particles from the soil surface, and ii) their subsequent transport downslope. Detachment is caused by raindrop impact and also by the abrasive power of surface runoff, especially when the runoff water flow has concentrated (Morgan, 2001). The downslope transport of detached particles occurs mainly with runoff water, to a lesser extent also by rainsplash. Soil erosion by water is highest for soils with a high percentage of silt and fine sand, e.g. loess soils (Schwertmann et al., 1987). Like runoff susceptibility, soil erodibility is enhanced by silting and crusting of the soil surface due to raindrop impact and splash during high-intensity rainfalls (Le Bissonais et al., 1995).

Numerous studies have been published on pesticide transport via surface runoff and erosion (e.g. White et al., 1976; Rohde et al., 1980; Haider, 1994; Klöppel et al., 1997; Lennartz et al., 1997; Spatz, 1999; Rübel, 1999; Wauchope et al., 1999; Louchart et al., 2001; Syversen, 2003). Pesticides lost in runoff and erosion events leave the field either dissolved in runoff water or adsorbed to eroded soil particles. However, for most pesticides losses via runoff are considered far more important than losses via erosion, because the amount of eroded soil lost from a field is usually small compared with the runoff volume (Leonard, 1990). Only for strongly sorbing substances with a Koc (Freundlich sorption coefficient normalized to soil organic carbon content) greater than ca. 1000 L kg-1, erosion is considered as the main loss pathway (Kenaga, 1980; Haider, 1994; Spatz, 1999). Compounds with intermediate sorption are more prone to being lost with surface runoff than weakly sorbing compounds, because the latter are quickly leached away from the soil surface by the infiltrating rainfall (Burgoa and Wauchope, 1995).

One measure to reduce pesticide inputs into surface waters via both runoff and erosion is the use of vegetated buffer strips (e.g. Popov et al., 2006) along field edges and water bodies. Also grassed waterways, which are frequently established in the US for erosion control, can reduce pesticide runoff and erosion inputs (Asmussen et al., 1977). A grassed waterway is basically a grassed buffer strip installed in up-and-down direction, with surface runoff from the upslope fields directed to it. Other possibilities for mitigating pesticide runoff and erosion inputs into surface waters are common measures to reduce surface runoff and erosion from the field, such as conservation tillage including zero-tillage (Fawcett et al., 1994), mulching, cover crops, contour ploughing/planting etc.. Also specific measures taken in vineyards to limit erosion (e.g. grass vegetation between vine rows) belong to this category. Constructed wetlands have also been proposed to mitigate the impact of pesticide runoff and erosion inputs (e.g. Moore et al., 2002). There are also mitigation options with respect to pesticide application, e.g. band spraying on row crops (Baker et al., 1995, cited by FOCUS, 2004b) or, if feasible, simply reduction of the application rate. This reduces the amount of pesticide that reaches the soil surface, and consequently also pesticide runoff and erosion losses from the field. Application as granules or incorporation of the pesticide into the soil are potential mitigation measures as well, yet are not applicable in all cases. The time passing between pesticide application and occurrence of a runoff event is also critical for runoff losses (Wauchope, 1978; Burgoa and Wauchope, 1995). Hence, avoiding application in seasons with a high probability of occurrence of runoff events (due to high-intensity rainstorms or saturated soils) would be another mitigation measure.

2.2 Drainflow

The purpose of installing artificial subsurface drains is to prevent topsoil saturation that otherwise would impair crop development, soil trafficability and workability. This excess water can either be due to shallow groundwater or slowly permeable horizons in the subsoil or an overall heavy texture. Consistent research findings have demonstrated that preferential flow phenomena are key contributors to the rapid transfer of pesticides to drainage systems (Kladivko et al., 1991; Harris and Catt, 1999; Funari et al., 1998; Novak et al, 2001; Accinelli et al., 2002). Preferential flow includes all phenomena where water and solutes move along certain pathways, while bypassing a fraction of the porous matrix (Hendrickx and Flury, 2001). It can be broadly distinguished into i) macropore flow along cracks, fissures, root channels and earthworm burrows (e.g. Schwartz et al., 1998; Flury et al., 1994), and ii) finger flow, which occurs in sandy soils (Ghodrati and Jury, 1990; Wang et al., 2003). For pesticide displacement in soils along preferential flow pathways the observation is characteristic that strongly adsorbing pesticides reach tile drains or lysimeter bottoms at the same time as mobile compounds; however, the amounts lost are still ranked according to the mobility characteristics of the pesticides (Flury, 1996). Pesticide transport by preferential flow to drains can cause high transient concentrations in agricultural ditches and rivers (Williams et al., 1996; Brown et al., 2004a). This is due to the fact that the relatively rapid movement of pesticide-loaded water through only a portion of the available pore space while bypassing a significant portion of the soil matrix decreases the residence time of the pesticide in the upper soil layers, where sorption is usually stronger and degradation is faster than in the subsoil. In other words, the infiltrating water does not have sufficient time to equilibrate with slowly moving resident water in the soil matrix (Jarvis, 1998). Although pesticide displacement by preferential flow was traditionally considered to be an issue restricted to heavy clay soils (Harris and Catt, 1999; Johnson and Haria, 1996), it has been demonstrated that it also plays an important role in lighter textured loamy or silty soils (Beven and Germann, 1982; Brown et al., 1995; Zehe and Flühler, 2001) and even occurs in poorly structured, homogeneous sandy soils (Hendrickx et al., 1993; Ghodrati and Jury, 1992). Yet, it is also evident from the literature that pesticide losses via drainflow are generally higher in heavy, structured soils than in sandy, weakly structured soils (Accinelli et al., 2002; Traub-Eberhard et al., 1995; DEFRA, 2003), unless the latter have a very shallow groundwater table. The main factors affecting pesticide inputs into surface waters via drainage are

- soil: texture, structure
- site: permeability of subsoil and vadose zone, depth of groundwater table
- drainage system: drain depth and spacing
- compound properties: sorption and degradation behaviour, volatility
- weather: temperature, rainfall distribution (especially the first weeks after application), to a lesser extent total amount of rainfall
- application rate
- application season: spring, summer or autumn (as for surface runoff, the time between application and the first drainflow event is critical).

Compared with runoff, there are relatively few possible mitigation measures for drainflow. Simple mitigation measures are reducing, dependent on the application season or not, the application rate or even imposing application restrictions on a) all drained soils or b) vulnerable drained soils, e.g. heavy clays (FOCUS, 2004b). Another option is shifting the pesticide application to an earlier (in autumn) or later date (in spring), when the soil is drier and less rainfall is to be expected. Furthermore, FOCUS (2004b) stated that many arable soils in Europe are over-drained (they didn't give references to confirm this assertion, though). As a consequence, the efficiency of the drains could be reduced to mitigate pesticide losses through drains (e.g. Harris et al., 2002). A further possible mitigation measure would be to establish collection ponds for tile drain outflow, in analogy to constructed wetlands for runoff mitigation. Also, creating a fine tilth of the topsoil has been proposed to reduce the generation of macropore flow and thus the transport of pesticides to drains (Brown et al., 2001).

2.3 Leaching

Leaching is vertical downward displacement of substances through the soil profile and the unsaturated zone, and finally to groundwater. Pesticide leaching is highest for weakly sorbing and/or persistent compounds, climates with high precipitation and low temperatures (which leads to high groundwater recharge) and in soils with either sandy texture and low organic matter (leaching by matrix flow) or soils exhibiting macropore flow, e.g. heavy loams and clays (see above). As leaching and drainage outflow of pesticides are similar processes, a lot of the points mentioned above in the drainage paragraph also apply to leaching. However, while drainflow is mostly a rather event-based process, leaching is usually more continuous in nature. This is mainly due to the typical kinds of soils where drainage and leaching predominate: Drained soils are usually fine-textured, clayey soils exhibiting a peaky, event-driven behaviour, while leaching to groundwater is often associated with somewhat lighter soils where matrix transport plays a more significant role.

A number of mitigation measures available for drainage can also be used for leaching: application restrictions for vulnerable soils and/or wet climates, reducing the application rate, and shifting the application to an earlier or later date. Also, creating a fine tilth of the topsoil or other tillage operations (e.g. conventional tillage instead of conservation or zero-tillage) to reduce macropore flow are possible measures to decrease leaching. To reduce pesticide leaching through the bulk soil ("matrix"), a possible mitigation measure is increasing the organic matter content of the soil by agronomic practices like incorporation of crop residues, in order to increase sorption of nonionic pesticides. Another option to reduce leaching by matrix flow would be switching to compounds with higher sorption and/or faster degradation (Flury, 1996).

2.4 Spray drift

During pesticide application by spraying, it is regularly observed that a certain portion of the applied amount is deposited outside the target area (Ganzelmeier et al., 1995), e.g. on soil, plant, and water surfaces. The extent of spray drift losses from the target area depends on weather conditions, technical equipment, application method, and target crop (Huber, 1998). Spraying on crops leads to higher drift than spraying on bare soil (FOCUS, 2004b). In contrast to the loss pathways mentioned above, spray drift losses are independent from the pesticide properties (but dependent on the formulation used). Simulations by Huber et al. (2000) and Röpke et al. (2004) suggested that total spray drift inputs into surface waters in Germany are much lower than inputs by surface runoff or drainage. Nevertheless, spray drift has, with respect to pesticide inputs into surface waters, been the main focus of most national pesticide regulation authorities (e.g. in Germany) for many years. This may be due to the fact that spray drift can lead to high, yet short-lived, levels of exposure in receiving water bodies.

As a consequence, the science of mitigation measures for pesticide exposure via spray drift is better developed than that for exposure via surface runoff or drainflow (FOCUS, 2004a). Mitigation measures for spray drift can be broadly divided into three classes (FOCUS, 2004a): i) the use of no-spray or even no-crop buffers, ii) the reduction of exposure using vegetative or artificial windbreaks, and iii) the application of drift-reducing technology. For iii), there are several different options: drift-reducing nozzles and spray additives to coarsen the droplet size distribution, shielded and band sprayers etc.

2.5 Other diffuse sources

Further diffuse input pathways for pesticides into surface waters are atmospheric deposition after volatilization and short-range or long-range atmospheric transport, and aeolian deposition of pesticide-loaded soil particles previously eroded by wind. For volatile pesticides modelling studies suggest that the former pathway, which is active on a longer range than spray drift, can be as important as spray drift (Loubet et al., 2006, Asman et al., 2003). However, there are only few possible mitigation measures available, e.g. spray additives, drift-reducing nozzles, and windbreaks. These are originally drift mitigation measures (see above), but should also have a side effect on volatilization / atmospheric deposition. Incorporating the pesticide into the soil to minimize volatilisation is another mitigation measure, yet this is applicable only in some cases. The latter pathway has importance only in areas where wind erosion is a problem. Pesticide input by wind erosion into surface waters can be mitigated by common measures for wind erosion control (e.g. windbreak hedges and ground cover). The two pesticide input pathways mentioned here will not be further discussed in the following.

2.6 Point sources

Point-source inputs of agricultural pesticides mainly consist of runoff from hard surfaces, mostly farmyards, storage facilities or roads. Typically the contamination of hard surfaces arises from filling and cleaning of sprayers, improper handling of tank mix leftovers, leaking of faulty equipment, incorrect storage of canisters (dripping from leaking or from insufficiently rinsed empty canisters) etc. (cf. Carter, 2000). Of course also accidental spills can occur, e.g. due to breaking or leaking tanks on the road to the field to be treated. There are two possible routes to surface water for pesticide runoff from farmyards: If the farmyard is not connected to the sewer system and there is no infiltration zone at the farmyard edge, the nearest surface water body will be the point of entry for the contaminated runoff water. If the farmyard is connected to the sewer system, pesticides will be transported to sewage plants. Since sewage plants are usually not fitted with active charcoal filters and degradation of pesticides usually does not occur to great extent in sewage plants (cf. Seel et al., 1994), pesticides tend to be released back into the environment through the sewage plant outlet. For Germany, it has been shown that at least in some regions point-source inputs contribute the majority to the observed pesticide loads in rivers (Müller et al., 2002; Fischer et al., 1998; Seel et al., 1996). For instance, Seel et al. (1996) found that in an intensively used agricultural region in Germany, two thirds of the pesticide load in the river originated from sewage plant outflows. Fischer et al. (1998) even found, for a small watershed (6.9 km²) in central Hesse, a contribution of point sources to the total pesticide load in the stream of more than 90 %. On an European level, a number of studies from a range of EU countries (Belgium, UK, Sweden, France, Germany) revealed a contribution of point sources to the total pesticide load in surface waters of 40-90 % (Jaeken and Debaer, 2005). No literature was identified on the EU-wide importance of point sources for groundwater contamination. However, a German study investigating the contamination sources for 6 pesticides frequently detected in groundwater (Dechet, 2005) revealed that of 181 examined and confirmed detections, 46 % classified as point sources according to our definition above, and only 32 % as diffuse sources.

One possible strategy to reduce the input from point sources is to increase awareness of the farmers with regard to pesticide handling and application, and to encourage them to implement loss-reducing measures (Kreuger and Nilsson, 2001). These measures of “best management practice” include filling and cleaning sprayers only on the field or on biobeds (Felgentreu and Bischoff, 2006), careful handling and storage of pesticides and safer storage of empty containers (Higginbotham, 2001), applying tank mix and container leftovers in dilute form on the field (Jaeken and Debaer, 2005), no application of pesticides on the farmyard etc. A further possibility would be to reduce the number of necessary sprayer filling and cleaning actions, which could be achieved by shared use of spraying equipment by farmers.

3 EFFECTIVENESS OF MITIGATION MEASURES AS INFLUENCED BY VARIOUS FACTORS

Roughly 180 publications directly dealing with or being somehow related to mitigation of pesticide inputs into water bodies were examined within the context of the present study. Both original studies and reviews were most numerous for the input path runoff and erosion (Table 1). However, not all experimental studies were usable for quantitative evaluation, e.g. because they did not contain quantitative estimates of reduction of pesticide load or surface water concentration by the respective mitigation measure. Some studies also did not deal with pesticides themselves, but with other agricultural contaminants like nitrate or phosphorus or just with water and sediment transport.

	input path				
	runoff/erosion	drainage	leaching	drift	point sources
original studies (experiments)	68	17	12	22	9
original studies usable for quantitative evaluation	27	4	0	14	5
reviews	19	4	2	6	1
other	1	1	2	4	4

Table 1. Number of publications examined dealing with or related to mitigation, separately for each input path (multiple counts possible).

3.1 Surface Runoff and Erosion

Classifying the large number of runoff studies according to the mitigation measure investigated revealed that the majority of experiments and reviews dealt with (vegetated) buffer strips (Table 2). Most of them were edge-of-field buffers directly below to a field, while only few studies investigated riparian buffers, i.e. buffers along the banks of streams or rivers. The context of most studies was improving surface water quality and/or improving process understanding.

	mitigation measures					
	buffer strips	constructed wetlands	grassed waterways	tillage practice	ground cover	other
original studies (experiments)	21 (edge-of-field), 5 (riparian)	6	3	3	2	2
original studies usable for quantitative evaluation	14 (edge-of-field), 2 (riparian)	4	2	2	1	1
reviews	10	5	1	1		1
other	1			1		1

Table 2. Number of studies investigated for the input path runoff and erosion (multiple counts possible).

Before presenting the evaluation of the results of the experimental studies, the findings of the examined existing reviews are summarized in chronological order. Several reviews focussed exclusively on buffer strips: Norris (1993), Muscutt et al. (1993), USDA (2000), Dosskey (2001), Lacas et al. (2005), Krutz et al. (2005), and Lovell and Sullivan (2006).

Norris (1993) concluded "The effectiveness of a buffer zone … evidently depends not only on its physical structure and on the kinds of pollutants which it must deal with, but also on its proximity to the source of pollution, simply because surface runoff must enter the buffer zone as shallow, overland flow, rather than already channelised streamflow. Making best use of the potential of buffer zones for protecting catchment water quality must therefore rely on their comprehensive arrangement over whole catchment areas."

Muscutt et al. (1993) remarked that stream water has diverse origins only some of which are likely to be affected by buffer zones. For example, concentrated surface flow through buffer zones owing to the occurrence of springs and ephemeral channels, or flow through subsurface drains, may affect buffer performance.

Baker and Mickelson (1994) found that results from buffer strip experiments were promising, but more research must be conducted under realistic field conditions. Moreover, they concluded that conservation tillage has the potential to reduce both runoff and erosion losses, and that pesticide incorporation into the soil is another way to reduce losses with surface runoff.

Fawcett et al. (1994) tried to quantify in their review the effect of conservation tillage on pesticide runoff to surface waters. They found that all three investigated conservation tillage systems (no-till, chisel ploughing, and ridge till) reduced herbicide runoff losses on average by 70, 69 and 42 %, respectively, compared with conventional tillage.

The USDA National Resources Conservation Service came to the conclusion that buffers to entrap and deposit sediment (and hence also strongly sorbing pesticides, which are mainly transported adsorbed to soil particles) are not required to be as wide as buffers used to remove soluble compounds such as nitrate or weakly or moderately sorbed pesticides (USDA, 2000), because it takes more surface area and longer flow paths to adsorb and infiltrate soluble material than to entrap solid material. Thus, USDA (2000) recommended buffer strip widths of at least 6 m for sediment and at least 30 m for dissolved compounds. Moreover, they stressed the need for buffer maintenance (removing sediment, mowing etc.) to uphold their functionality. They also remarked that "concentrated flow is the nemesis of pesticide trapping by buffers", but can be re-dispersed to sheet flow by innovative technology such as level spreaders, water bars and stiff-grass hedges.

Dosskey (2001) concluded in his review that it remains unclear what degree of pollution reduction is to be expected from converting some of the farmers' cultivated land to buffers. Furthermore, he did not find any studies that reported on the impact of riparian buffer installation on pollutant levels in streams or lakes. He summarized that "Buffer performance is greatest when runoff flows across a buffer in shallow uniform (sheet) flow. Uneven land that concentrates runoff flow within a buffer can substantially limit buffer effectiveness."

The FOCUS Working Group on landscape and mitigation factors in ecological risk assessment stated (FOCUS, 2004b): "The main question is not really the identification of the mechanisms involved, which are quite well known, but their quantification and their relative predominance." With respect to buffer strips, they acknowledged that hydraulic by-passes (rills, gullies, ditches, tile drains) through the buffer zone can totally invalidate their effectiveness, and that the occurrence of concentrated flows is more likely along streams (riparian buffers) than uphill (edge-of-field buffers). Furthermore, both FOCUS (2004b) and ECOFRAM (1999) present a "mitigation practices summary guide" table, based upon experience in managing runoff in the USA (SETAC, 1994). This table gives potential reduction efficiencies of various mitigation measures for runoff losses to surface water. However, the efficiency ranges given there are rather wide (e.g. 20-60 % for the retention of strongly sorbed pesticides in vegetative filter strips, or 20-90 % for strongly sorbed pesticides in constructed wetlands), which hampers the direct use of these reduction efficiencies for modelling purposes.

Schulz (2004) compiled the results of studies in which diffuse-source insecticide inputs, resulting from normal farming practice, were measured in aquatic ecosystems. Based on rainfall-runoff relationships derived by Lutz (1984) and Maniak (1992), he questioned the suitability of buffer strips to retain dissolved pesticides. He argued that heavy rainfall events causing storm runoff are always associated with the production of very large water volumes in a short time, which in many cases will not be retained by any sort of widely employed buffer strip ("hydrological dilemma"). For a more effective mitigation of diffuse pesticide inputs into surface waters, Schulz (2004) suggested the use of constructed wetlands or vegetated ditches, as the available experimental efficiencies from 9 studies were very promising (load reductions between 54 % and > 99.9 %, and in most cases > 90 %).

Lacas et al. (2005) stated that, although the main processes and properties of the strips which determine their interception effectiveness are known at least from a qualitative point of view, the prediction of the interception effectiveness of a given strip still seems unattainable with the present state of knowledge. They identified two main reasons for this: i) The number of interacting processes and strip properties is so large that the global functioning of a strip does not seem to be predictable by a simple model and from a few characteristics of the strip (e.g. length). ii) Some processes are still insufficiently described from a quantitative point of view, e.g. the channelling of surface flow within a strip, the fate of fine solid particles with respect to sedimentation and infiltration, the adsorption on soil and plant materials, and the temporal changes in strip characteristics due to biological activity and/or the sedimentation process. Major points to be studied according to Lacas et al. (2005) are the fate of degradation metabolites in the buffer strips and the impact of subsurface flow on the global effectiveness of buffer systems, especially of riparian buffer strips. Furthermore, they stressed the need for developing physically-based models for buffer strips, in order to improve the predictability of their effectiveness.

Krutz et al. (2005) examined in their review the factors reported in the literature to affect the retention of herbicides in vegetative buffer strips. For instance, they found that in general, the effectiveness of a buffer strip indeed increases with strip width (except for strongly sorbing compounds, which are transported with sediment and are often deposited already after a short flow distance in the strip). In contrast, the area ratio of source area to buffer strip mostly did not significantly influence buffer effectiveness in the reported area ratio range (5:1 to 45:1). Furthermore, Krutz et al. (2005) identified a negative correlation between antecedent soil moisture content and herbicide retention (two studies), and an increase of relative retention with nominal inflow concentration (one study, cf. Misra et al., 1996), probably due to adsorption. They felt that this concentration dependence may invalidate a comparison of the retention of different pesticides.

Finally, Lovell and Sullivan (2006) identified the lack of knowledge on the effect and effectiveness of buffers at watershed scale as one of the reasons why buffers are still underused in US agroecosystems.

Our examination of the available original literature yielded 14 publications on edge-of-field buffer strips suitable for a quantitative evaluation of the pesticide load reduction efficiency of the strips (Table 3). From the compiled studies the following can be summarized:

reference	country	buffer strip character-istics	source area character-istics	area ratio (source/strip)	soil type (texture)	method of runoff generation	time scale of experiment / design of simulated runoff study	efficiency (load reduction) in %	remarks
Arora et al. (1993)	USA	grassed; w = 20.1 m [1)], l = 1.5 m [1)]	0.41 ha	15 / 30 (ratios achieved by inflow regulation)	silt loam	natural rainfall used to simulate run-on	1 event; 6 strips, n = 3 [2)]	runoff: 13.1 / 3.9 [3)] sediment: 45.7 / 40.6 atrazine: 12.5 / 9.3 metolachlor 27.3 / 15.3 cyanazine 21.1 / 7.2	tank between field and strips for collection and redistribution; 1 event analyzed
Arora et al. (1996)	USA	grassed; w = 20.1 m, l = 1.5 m	0.41 ha	15 / 30 (ratios achieved by inflow regulation)	silty clay loam	natural rainfall used to simulate run-on	2 years 6 strips, n = 3	sediment: 40-100 atrazine: 11-100 metolachlor: 16-100 cyanazine: 8-100	same experiment as in Arora et al. (1993) though different soil is stated; variation refers to area ratios and the 6 different events which were fully characterized
Klöppel et al. (1997)	Germany	grassed; w = 10 / 15 / 20 m, l = 10 m	-	-	silt loam	simulated rainfall + run-on	7 variants, n = 1	runoff: 0-92 terbuthylazine: 70-98 isoproturon: 70-98 dichlorprop-P: 61-98	slightly more load reduction (ca. 15 %) in 20 / 15 m than in 10 m strips
Krutz et al. (2003)	USA	grassed; w = 3 m, l = 1 m	-	30 (virtual)	clay (Vertisol)	simulated run-on	n = 8 (4 replicates in each of 2 years)	atrazine: 22 metabolites: 18-20	buffer saturated before start of experiment; adsorption plays a role, but less than infiltration (atrazine: 40/60)
Misra et al. (1996)	USA	grassed ; w=12.2 m, l = 1.5 m	-	15 / 30 (achieved by inflow regulation)	loam	simulated rainfall + run-on	12 strips, n = 3	atrazine 41 / 37 metolachlor 39 / 35 cyanazine 38 / 34	decrease of removal with increasing area ratio not significant; increase of removal with increasing inflow conc. significant

Popov et al. (2005)	Australia	grassed ; w = 4 m, l = 1.25 m	-	-	clay (Vertisol)	simulated run-on (20-800 mm)	14 strips, n = 2	runoff: 39-74 sediment: 57-93 atrazine: 40-85 metolachlor: 44-85	variation refers to 7 different treatments; > 160 mm run-on depth: only infiltration effective; < 80 mm: significant herbicide adsorption
Schmitt et al. (1999)	USA	different vegetation types; w = 7.5 / 15 m, l = 3 m	-	10.8 / 5.4 (virtual)	silty clay loam + sandy loam (gradient)	simulated rainfall + run-on	40 strips, n = 5	runoff: 36-82 sediment: 80-99 permethrin: 47-97 atrazine: 33-90 alachlor: 42-93	doubled strip width increased infiltration and dilution substantially, but not sedimentation
Syversen (2003)	Norway	grassed; w = 5 m, l = 10 m	w = 45 m, l = 10 m	9	silty clay loam	natural rainfall	3 years, n = 2	sediment: 51 glyphosate: 48 propiconazole: 85 fenpropimorph: 34 AMPA: 67	experimental period differed between compounds
Syversen and Bechmann (2003)	Norway	grassed; w = 5 m, l = 5 / 7.5 m	-	-	silty clay loam	simulated run-on	1 strip, 4 events	sediment 62 glyphosate 39 propiconazole 63 fenpropimorph 71	lower removal efficiency for glyphosate probably due to adsorption to fine clay particles which are less deposited than other particles
Patty et al. (1997)	France	grassed; w = 6 / 12 / 18 m, l = 5 m	w = 50 m; l = 5 m	8.3, 4.2, 2.8	silt loam	natural rainfall	1 (2) years	runoff: 43-99.9 sediment: 87-100 lindane: 72-100 atrazine: 44-100 isoproturon: > 99 diflufenican: > 97	3 different study sites: IPU and diflufenican at 1 site, lindane and atrazine at the 2 others; efficiency either very high throughout or increasing with strip width
Rankins et al. (2001)	USA	grassed : w = 0.3 m, l = 4 m	w = 22 m , l = 4 m	73.3	silty clay (Vertisol)	natural and simulated rainfall	3 years as replicates	runoff: 46-76 sediment: 66-80 fluometuron : 59-84 norflurazon: 45-86	variation refers to different grass species studied

Tingle et al. (1998)	USA	grassed; w = 0.5-4 m, l = 4 m	w = 22 m , l = 4 m	5.5-44	silty clay (Vertisol)	simulated rainfall	3 years as replicates, results for 2 and 84 days after app.	runoff: 47-93 sediment: 82-98 metolachlor: 67-97 metribuzine: 73-98	variation refers to different strip widths and time periods between application and event; no significant effect of filter strip width
Webster and Shaw (1996)	USA	grassed; w = 2 m, l = 4 m	w = 22 m , l = 4 m	11	silty clay (Vertisol)	natural and simulated rainfall	3 years as replicates	runoff: 14-47 metolachlor: 39-64 metribuzine: 41-64	same study site as for Tingle et al. (1998) and Rankins et al. (2001); variation refers to 3 different cropping systems
Spatz (1999)	Germany	grassed, w = 1, 4, 5, 7, 10, 15 m, l = 0.6 m	w = 7 m, l = 0.6 m	7-0.47	silt loam + silty clay loam (2 sites)	simulated rainfall (not on buffers)	12 treatment variants	runoff: 4-99 sediment: 72-98 pirimicarb: 10-100 mecoprop: 0-99 isoproturon: 2-99 terbuthylazine: 17-99 fenpropimorph:35-100 pendimethalin: 72-100	efficiency increased with increasing strip length and decreasing soil moisture; irrigation of strips caused pesticide remobilisation; variation refers to different strip lengths and different treatments (rainfall intensity and duration, initial soil moisture in the strip and in the source area)
Spatz (1999)	Germany	different types, w = 5 / 10 / 20 m	n. def.	n. def.	silt loam	natural rainfall	1 cropping season (1994)	runoff: 0-100 sediment: 0-100 terbuthylazine (aq [4)]): 72 (0-100) terbuthylazine (sed [4)]): 83 (13-100) pendimethalin (aq): 53 (0-100) pendimethalin (sed): 76 (0-100)	1 extreme event caused permanent gullies and > 75 % of all pesticide losses in the season; percentage transported in water phase: terbuthylazine 36, pendimethalin 2; variation refers to different buffer types (grass, mulch, crop, fallow), events and strip widths

Spatz (1999)	Germany	different types, w = 7 / 15 m, l = 2.5 m	w = 20 m, l = 2.5 m	2.86 / 1.33	silt loam	natural rainfall	1 cropping season (1995)	runoff: 0-96 sediment: 0-100 terbuthylazine (aq): 76 (33-100) terbuthylazine (sed): 98 (94-100) isoproturon (aq): 92 (50-100) pendimethalin (aq): 74 (65-84) pendimethalin (sed): 54 (0-100; for grassed strips: 100)	only 4 small runoff events in this season; percentage transported in water phase: terbuthylazine 98, isoproturon 100, pendimethalin 30; IPU was applied to barley plots, terbuthylazine and pendimethalin to maize; variation for pesticides refers to different buffer types (grass, crop, fallow), events and strip widths

[1)] w = width: dimension up-and-down; l = length: dimension perpendicular to the slope

[2)] n = number of replicates

[3)] for area ratios of 15 and 30, respectively

[4)] aq = dissolved in water, sed = adsorbed to sediment particles

Table 3. Field studies on the effectiveness of edge-of-field buffer strips in reducing pesticide runoff and erosion losses.

- The available studies were carried out by only a limited number of research groups (9) and only on a limited range of soil textures (mostly silty).
- The experimental designs (plot setup, strip and source area, generation of runoff, number of runoff events and replicates etc.) differed considerably between the studies.
- Load reduction efficiency was sometimes obtained by comparison between strip inflow and outflow, and sometimes by comparison of strip outflow with a control without buffer strip. The latter methodology suffers from variability between source areas (e.g. of soil properties).
- 4 of the 10 studies were performed on Vertisols, which crack deeply upon drying and are thus very prone to macropore flow. These studies might thus overestimate the pesticide load reduction efficiencies of the vegetative buffer strips, unless the buffer strip was saturated at the start of the experiment (as in Krutz et al., 2003).
- In a range of studies employing simulated rainfall or run-on, buffer strips were irrigated neither before nor during the experiment and were therefore dry at the beginning of runoff inflow into the strip (Rankins et al., 2001; in part: Spatz, 1999; Tingle et al., 1998; Webster and Shaw, 1996). This possibly represents best-case rather than average-case conditions and thus leads to too optimistic infiltration rates and buffer strip efficiencies.
- Grassed strips were more effective than strips with crop or bare soil in reducing loss of sediment and sediment-bound pesticides (Schmitt et al., 1999; Spatz, 1999). A higher efficiency of grassed strips in reducing runoff volumes and dissolved pesticide loads could not be established.
- Load reduction efficiency was not substantially different between weakly and moderately sorbing pesticides. Results for strongly sorbing pesticides that are mainly transported in the sediment phase, e.g. pendimethalin, are scarce.
- Efficiency depends on the nature of the runoff event (Arora et al., 1996; Spatz, 1999). For instance, the study of Arora et al. (1996) demonstrated a strong impact of the temporal variability of runoff events (due to different rainfall amounts and intensities and different antecedent soil moisture conditions) on the effectiveness of buffer strips: Some smaller runoff flows (and the associated pesticides) infiltrated completely, whereas some large runoff flows were not significantly retained by the strips.
- The reduction in pesticide load was mainly due to infiltration and sedimentation in the buffer strip. In some cases, however, also significant adsorption to plant or soil material in the strip occurred (Krutz et al., 2003; Misra et al., 1996; Popov et al., 2005; Spatz, 1999 (only pendimethalin)). In this context, Popov et al. (2005) stated that “small plots, or at least high flow rates associated with small plot studies, appear likely to underestimate the reduction in concentrations that can occur under practical field conditions. This may

partly explain why the literature generally suggests that the main benefit of vegetated filter strips arises from infiltration". They further remarked that the small plots used in their study led to an overestimation of infiltration due to border effects (lateral flow), and that this overestimation may be a problem of many plot studies reporting high efficiency of biofilters.

- A significant effect of the buffer strip width on pesticide trapping efficiencies was not observed in all studies. However, this does not seem illogical, as the strip width should not matter much when infiltration/sedimentation need less flow length than the shortest strip or several times more flow length than the longest strip.

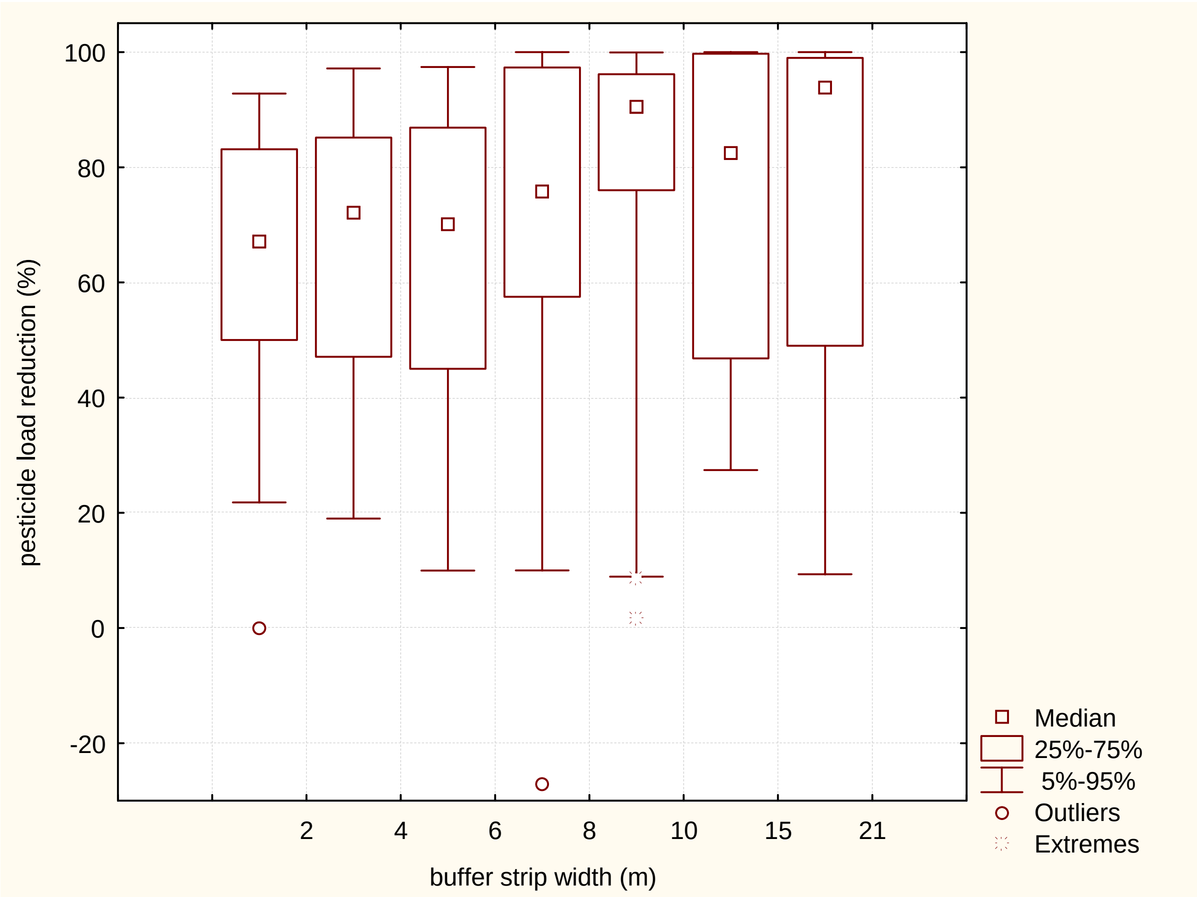

Fig. 1: Pesticide load reduction efficiencies of edge-of-field buffer strips vs. classified buffer strip width for the studies in Table 3.

The data points from each study refer to single pesticides and are averaged over replicates (if present) and observation periods (if applicable). Variability between different treatments (source/strip area ratios, simulated rainfall/run-on regimes, strip vegetation types etc.) was preserved as much as the reported study data allowed. All compounds used were included in the plot. Number of data points: 277. Negative efficiencies arise from variability between the source areas of buffer strip and strip-free control.

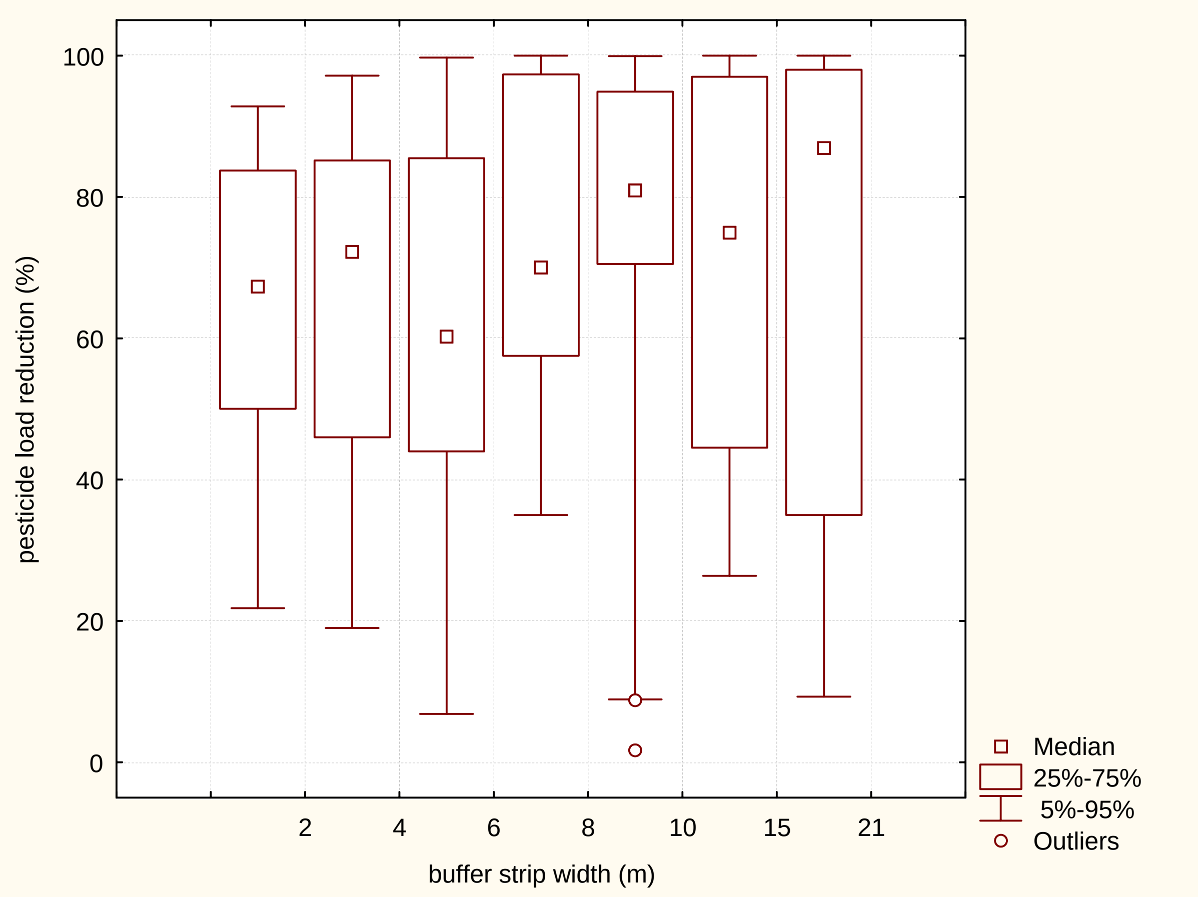

Fig. 2: Pesticide load reduction efficiencies of edge-of-field buffer strips vs. classified buffer strip width for the studies in Table 3.

Only compounds predominantly transported in the water phase (K_{oc} < 1000 L kg^{-1}) or separately analysed in the water phase were included. Number of datapoints: 214.

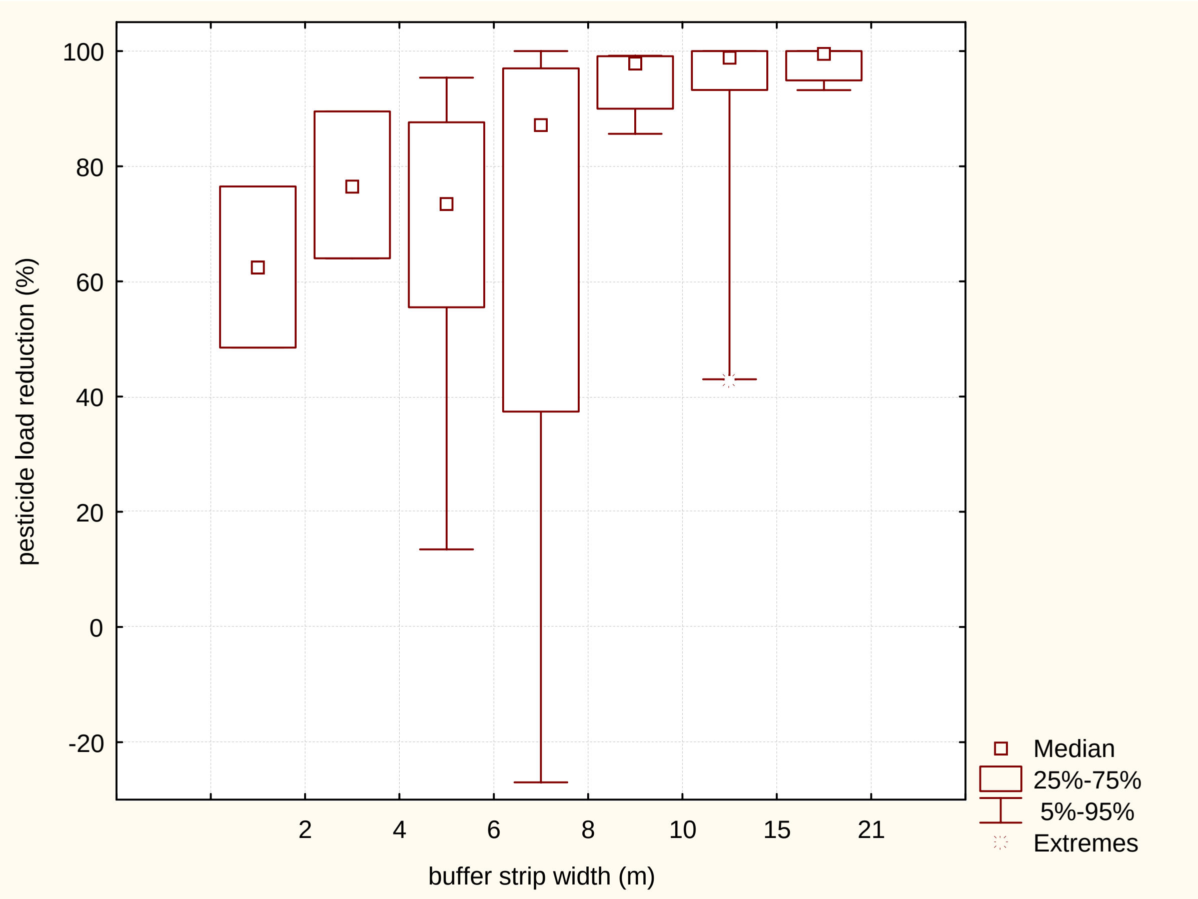

Fig. 3: Pesticide load reduction efficiencies of edge-of-field buffer strips vs. classified buffer strip width for the studies in Table 3.

Only compounds predominantly transported in the eroded sediment phase (K_{oc} > 1000 L kg^{-1}) or separately analysed in the sediment phase were included. Number of datapoints: 63. Negative efficiencies arise from variability between the source areas of buffer strip and strip-free control.

It can be concluded from the examined studies (Table 3) and reviews that **the effectiveness of grassed buffer strips located at the lower edges of fields in reducing pesticide runoff and erosion losses has been demonstrated in general**. However, it is also obvious that this effectiveness is very variable and that this variability cannot be explained by strip width alone (Fig. 1). It is therefore difficult to derive recommended efficiency values for modelling purposes. At least for buffer strip widths greater than 8 m, load reduction efficiencies were in tendency larger for pesticides with the major portion transported in the sediment phase (Fig. 3) than for pesticides predominantly transported in the water phase (Fig. 2). On average, the pesticide load reduction efficiencies observed in the examined studies are roughly comparable to those assumed by the German regulatory model EXPOSIT 1.1 (Winkler, 2001), which are 50 % reduction for 5 m buffer strip width, 90 % for 10 m width, and 97.5 % for 20 m width and apply to pesticides as well as runoff volume and sediment (it should be noted here that EXPOSIT does not differentiate between dissolved and particle-bound pesticide transport). Hence, the reduction efficiencies proposed in EXPOSIT seem defensible for modelling purposes with respect to edge-of-field grassed buffer strips. For unfavourable conditions (e.g. large runoff/erosion events, wet antecedent soil moisture condition of the strips, large source/strip area ratios) though, the EXPOSIT values may considerably overestimate the effectiveness of edge-of-field buffers and substantially lower efficiency values may be necessary to avoid an underestimation of risk.

As already pointed out in some of the reviews above, it must be clearly distinguished here between buffer strips directly adjacent to the field at its lower edge, and riparian buffers, i.e. bank vegetation along streams and rivers. Only two publications with quantitative results for pesticide load reduction by riparian buffers were identified. Moreover, these two publications were companion papers, with the studies conducted at the same location and in the same time period (Lowrance et al., 1997; Vellidis et al., 2002). The riparian buffers in the two studies (a mature, managed woodland and a newly restored woodland) achieved almost complete pesticide retention. However, with 50 and 38 m width, respectively, the buffers were very wide. Such wide riparian buffers are only rarely present in intensively used European agricultural landscapes, and their installation would require setting aside a lot of crop land. Other studies suggested that bank vegetation along surface water bodies is rather ineffective in reducing chemical inputs via runoff and erosion. Parsons et al. (1995) observed that the resistance of natural bank vegetation to surface runoff entering the strip as concentrated flow was very low. Bach et al. (1994) demonstrated for a typical German low mountain agricultural area that only 1 to 6 % of the river length adjacent to agricultural fields possessed *functional* bank vegetation filter strips against pesticide runoff and erosion inputs. The main reasons for this were that i) surface runoff entered the bank vegetation strips mainly as concentrated flow (as opposed to laminar sheet flow), which greatly diminishes the filter efficiency, and ii) most bank vegetation strips were not suitable to effectively reduce pesticide runoff and erosion inputs even for sheet flow, because they were either too narrow or too sparsely vegetated. Fabis et al. (1994) found in a Molybdenum tracer experiment that 32-90 % of solutes that had infiltrated into 4.5-20 m wide bank vegetation filter strips nevertheless reached the stream via rapid interflow. Also, even if pesticide-loaded runoff infiltrates into a riparian buffer strip, the groundwater table below the strip will be rather shallow (unless the stream bed is deeply cut into the floodplain), and the groundwater feeds into the nearby stream. On basis of the results of the examined studies and the reviews above, we conclude that **riparian buffer strips are most probably much less effective than edge-of-field buffer strips**. We therefore join the opinion of Gril and Lacas (2006), who stated that the importance of riparian buffer strips is higher for drift control than for runoff control.

With respect to constructed wetlands, no other studies with quantitative results were identified than those already cited and discussed by Schulz (2004) and FOCUS (2004b). The vast majority of these studies (e.g. Schulz and Peall, 2001) suggest that constructed wetlands are very effective in reducing pesticide inputs into surface waters. A potential drawback is that they can be quite area-consuming: the largest investigated wetland was 134 m long and 36 m wide (Moore et al., 2002). However, smaller, less area-demanding wetlands (e.g. 50 m long and 1.5 m wide; Moore et al., 2001) have been found to be very effective in removing pesticides from the water passing through the wetland. Yet, it has to be noted that almost all available studies dealt with strongly sorbing insecticides (e.g. chlorpyrifos) with a strong tendency to adsorb to macrophytes, suspended particles or bed sediment. Only one study (Moore et al., 2000) investigated fate and transport of the moderately sorbing herbicide atrazine in constructed wetlands. Moore et al. (2000) found that a travel distance of 100-280 m through the wetland would be necessary to achieve an effective runoff mitigation (more precisely: an atrazine concentration in outflow corresponding to the NOEC for higher aquatic plants). It can therefore be concluded that more research on the effectiveness of constructed wetlands for removing moderately and weakly sorbing pesticides must be conducted.

Only two studies were found on the effectiveness of grassed waterways: Rohde et al. (1980) and Asmussen et al. (1977). Both studies were conducted on the same grassed waterway with a flow length of 24.4 m. Rohde et al. (1980) found total load reductions of 96 % (dry) and 86 % (wet antecedent soil moisture condition) for the strongly sorbing herbicide trifluralin. 43 % (dry) and 29 % (wet) of the load reduction were attributed to infiltration. Asmussen et al. (1977) observed a reduction of runoff volume by 50 % (dry) and 2 % (wet), and of sediment load by 98 % (dry) and 94 % (wet antecedent soil moisture condition). The load of the weakly sorbing herbicide 2,4-D, which was almost completely transported in the water phase, was reduced in the waterway by 72 (dry) and 69 % (wet). The higher reduction efficiency for 2,4-D compared with runoff water was attributed to interactions of 2,4-D with the grass cover. Although the two studies gave promising results and suggest high removal efficiencies of grassed waterways, the available data base is very small. However, the results for grassed buffer strips, which are physically similar to grassed waterways, corroborate the results of the two studies. It should be noted here that grassed waterways are not a commonly applied practice in European agriculture yet, and that, as a consequence, for their establishment cropped land has to be set aside.

Two studies with quantitative results investigating the effect of tillage practice on pesticide runoff and erosion losses were examined. Isensee and Sadeghi (1993) found that the effect of tillage practice on the volume of runoff from a silt loam soil was dependent on the antecedent soil moisture condition: When the soil moisture content was high (6 or less days after the last rainfall), runoff was higher from no-till plots than from conventional-till maize plots. The reverse was true when the runoff event occurred at 7 or more days after the last rainfall event. Atrazine and cyanazine concentrations in surface runoff were significantly higher for no-till than for conventional-till plots, and consequently annual atrazine and cyanazine losses were higher by a factor of 1.5-3 from no-till than from conventional-till plots. Losses of alachlor, which was the only compound applied in microencapsulated form, were much lower than for the other herbicides and were not affected by tillage practice. The results of Isensee and Sadeghi (1993) contradict the intuitive assumption that zero-tillage should reduce pesticide runoff losses compared with conventional tillage. However, Fawcett et al. (1994) cite several studies that indeed yielded lower runoff losses from zero-tillage than under conventional tillage, e.g. Hall et al. (1991) and Hall et al. (1984). This inconsistency of experimental results suggests that further factors, e.g. subsoil permeability, influence the effect of zero-tillage on pesticide runoff losses. Kenimer et al. (1997) reported that runoff losses of alachlor and terbufos from contour-tilled plots on a silt loam soil were lower by factors of 2.6 and 1.25, respectively, in comparison to up-and-down-tilled plots.

Sadeghi and Isensee (2001) investigated the effects of ground cover on pesticide runoff losses. They found only slightly, non-significantly lower surface runoff and atrazine and metolachlor losses from no-till corn plots with a vetch cover crop residue compared with no-till plots without vetch residue. Gril et al. (1989) tested several types of ground cover for their effectiveness in limiting runoff and sheet erosion in vineyards. Best performance was achieved with permanent grass-sodding between rows. Yet, their study did not include pesticides. The scarcity of available data clearly shows that there is a need for more research on the effectiveness of vegetative ground cover in reducing pesticide runoff and erosion losses.

The effect of pesticide formulation on runoff losses was investigated by Kenimer et al. (1997). They found higher sediment-borne and total losses of microencapsulated alachlor compared with alachlor applied as emulsifiable concentrate, and no significant differences in terbufos losses between controlled-release and granular formulation. Again, more experimental research is needed to broaden the data base.

Kladivko et al. (2001; see section 3.2) found for USA and Southern Canada that in general the presence of subsurface drainage decreases surface runoff losses of reactive compounds such as pesticides, both because of lower runoff volumes and often also because of lower concentrations in runoff water due to the delayed initiation of surface runoff. Moreover, they found that pesticide concentrations and mass losses were, under North American conditions, usually much lower in subsurface drainage than in surface runoff, often by an order of magnitude. These results suggest that subsurface drainage could be viewed as a further mitigation measure for pesticide runoff losses. The findings of Brown et al. (1995) for a clay loam soil in NE-England confirm this perception: They found that total losses of autumn-applied pesticides from an undrained plot were up to 4 times larger than from a mole-drained plot. The reason was that the mole drains reduced the amount of surface layer flow (surface runoff + shallow interflow) in this slowly permeable soil prone to waterlogging. It can be concluded that subsurface drains are an effective mitigation measure for slowly permeable soils with frequent waterlogging (however, such soils mostly cannot be used for arable farming without subsurface drainage anyway). As, of course, the soil should not be over-drained either (see below), a compromise between runoff and drainage losses has to be found to minimize total loss.

3.2 Drainage and leaching

Compared with surface runoff and erosion, there is considerably less literature available on mitigation of pesticide losses via drainflow and leaching (cf. Table 1). In his extensive review on pesticide transport through field soils, Flury (1996) found, using studies in North America and Europe, that:

- The mass lost by drainage or leaching seems, in general, to be smaller than that lost by runoff.
- Conservation tillage (incl. zero tillage) had either no effect on pesticide leaching / drainflow or enhanced it compared with conventional tillage.
- The experimental evidence on the effect of pesticide formulation is not consistent. Controlled-release formulations may reduce the risk of pesticide transport by preferential flow, but might increase slow leaching at later times. Granular formulations yield a less uniform spatial distribution of pesticides at the soil surface than sprayable formulations, which also may affect the transport of active ingredient.

Furthermore, Flury (1996) argued that "Leaching and [surface] runoff are mutually dependent processes. During runoff, a portion of the water moves laterally to surface waters, and does not contribute to leaching any more. Increased runoff is therefore related to decreased leaching. This might not be generally true for preferential flow processes through macropores, but certainly for leaching of chemicals through the bulk soil." Hence, steps to reduce surface runoff would inevitably lead to a potentially higher risk of loss through leaching.

This statement of Flury (1996) can however be questioned for most European conditions, since even on runoff-prone soils runoff usually does not dominate the water balance. Only when surface runoff holds a significant share of the annual water balance, changes in surface runoff will lead to significant changes in percolation and pesticide leaching.

Kladivko et al. (2001) reviewed the results of more than 30 North American studies of pesticide transport to subsurface drains. They concluded that pesticide losses via drainflow are not a major problem in the USA and recommended to place the highest priority on managing surface runoff losses of pesticides (see above). However, pesticide application in the examined studies mainly took place in spring giving only a short period of drainflow before the onset of summer water deficits. It should be noted that these conclusions drawn in the US are unlikely to be transferable to Europe because of differences in climatic conditions (especially water balance and rainfall patterns), agricultural practice, drainage characteristics or soil properties.

A review conducted by DEFRA (2003) for drainflow studies performed in Europe revealed decreasing seasonal drainage losses and maximum concentrations in drainflow with increasing sand content (i.e. decreasing macropore flow). DEFRA also concluded that no-tillage practices either have no effect on pesticide losses to drains or yield higher losses compared to conventional ploughing. With respect to pesticide formulations, they cited a study (Schreiber et al., 1993) where maximum concentrations of atrazine in drainflow were reduced by 50-80 % for plots treated with a starch-encapsulated formulation. However, the results of Brown et al. (1995) were contradictory: The authors reported unexpectedly high drainage losses of fonofos and suggested enhanced macropore displacement due to the formulation as microcapsules as a possible explanation.

FOCUS (2004b) mainly repeated the results of Kladivko et al. (2001) and DEFRA (2003). The group furthermore acknowledged the suitability of soil structure management, avoiding application to very dry or very wet soils, and discouraging the practice of over-draining, but stated that none of these mitigation measures is suitable for inclusion in ecological risk assessment as their “impact on pesticide transport is unpredictable and none can be rigorously controlled or policed” (FOCUS, 2004a).

The available original literature yielded only few usable studies with respect to mitigation of drainage inputs. In a lysimeter experiment, Brown et al. (2001) found that generation of a fine topsoil tilth prior to application reduced isoproturon drainage losses by ca. 35 % over the monitoring period in mole-drained lysimeters of a heavy clay soil compared with standard tilth. A similar experiment (Brown et al., 1999) using the same soil type even yielded three times lower isoproturon drainage losses for the fine tilth treatment over the monitoring period. However, it can be expected that for soils less prone to macropore flow the effect of a fine topsoil tilth will be considerable less. Moreover, for particle-bound compounds a fine topsoil tilth or other intensive tillage operations might have the opposite effect and increase losses (Jarvis and Dubus, 2006).

A trial in the same heavy clay soil at Brimstone Farm (Jones et al., 1995) suggested that incorporation of pesticide into the topsoil following application had no effect on subsequent losses to drainflow. In contrast, Gish et al. (1991) reported that soil-incorporated carbofuran leached less than atrazine and cyanazine, which were applied as surface broadcast sprays, despite a much larger inherent mobility of carbofuran.

Harris et al. (2002) found that reducing drain efficiency led to an average reduction of isoproturon drainage loss by 30 %, overall drainflow by ca. 20 % and peak drainflow by 10 % at Brimstone. Yet, a field study on a clay loam in Northumberland (Brown et al., 1995) showed that losses of four pesticides in surface runoff plus shallow interflow from an undrained plot were up to four times larger than combined losses in surface runoff, shallow interflow and drainflow from an adjacent plot with mole drains. Moreover, reducing drain efficiency could in some cases enhance pesticide leaching to groundwater (due to increased percolation through the bottom of the soil profile). Hence, if reducing drain efficiency is really to be used as a mitigation measure for drainage losses, it will have to be handled very carefully.

No suitable studies on mitigation measures for pesticide leaching were identified.

In summary, the limited amount of available literature suggests that the effects of pesticide formulation, tillage operations and pesticide incorporation into the soil on pesticide losses via drainage and leaching are insufficiently known and at best unpredictable. These measures are therefore not suitable for recommendation as mitigation measures for pesticide losses via drainage or leaching. This leaves product substitution, application rate reduction and shift of the application date as only feasible mitigation measures for both pathways. For drainage, the use of collection ponds for drain outflow analogously to constructed wetlands (see section 3.1) seems a further possible alternative, but there are no experimental data available so far on their effectiveness.

3.3 Spray drift

There is a considerable amount of literature available on mitigation of drift inputs into surface waters (Table 1). An exhaustive review on the various factors influencing drift and on possible mitigation measures (three classes: no-spray buffers, windbreaks, drift-reducing technology) has been compiled by the FOCUS Working Group on landscape and mitigation factors in ecological risk assessment (FOCUS, 2004b). Their conclusions and key references are summarized briefly below (for further information, the reader is invited to consult the FOCUS document):

- There is a strong positive correlation between wind speed and spray drift deposition (Arvidsson, 1997).
- The experimental results on the effect of sprayed crop type are not consistent. Ganzelmeier et al. (1995) found only minor differences in spray drift when spraying on cereals and bare soil. In contrast, Stallinga et al. (1999) found larger drift for cereals than

for bare soil, but no differences between different crop heights. Finally, Van de Zande et al. (2003) found differences of spray drift for different crop types.

- Crop-free no-spray buffer zones are effective in reducing spray drift inputs into surface water bodies (Porskamp et al., 1995). Moreover, mitigation may be simpler to enforce where no-spray buffers are legislated as no-crop buffers, as in the Netherlands, because the spray operator has no reasons to spray over a no-crop zone.
- Spray drift deposition beyond crop-free no-spray buffers decreases with increasing height of the vegetation in the buffer strip (e.g. Van de Zande et al., 2000a).
- Spray drift deposition on vegetation differs from deposition on the ground, and dose response from spray application (i.e., direct overspray) is different from dose response from drift deposition (Koch et al., 2002).
- Spray drift deposition on ditch water surfaces depends on the layout of the ditch (Porskamp et al., 1995), e.g. slope and width of banks, and height of water table relative to field level.
- Spray drift increases with driving speed of the sprayer (Arvidsson, 1997).
- Spray drift increases with sprayer boom height (De Jong et al., 2000; Arvidsson, 1997).
- Reducing sprayer boom height increases the drift reduction efficiency of air assistance (Van de Zande et al., 2000b).
- The coarser the spray quality, the lower the spray drift. A coarser spray quality can be achieved by many ways: nozzle type (up to 90 % drift reduction efficiency), air assistance (> 50 %), or tank additives (20-50 %).
- Low-drift nozzles differ in their effectiveness and must be ranked and classified according to their level of drift reduction compared with a standard nozzle.
- Band sprayers can reduce drift by 90 % compared with standard field sprayers (Van de Zande et al., 2000c).
- Special “end nozzles” that are mounted on the end of the spray boom and produce a cut-off fan spray are a further drift mitigation measure.
- A shielded and a tunnel sprayer were found to reduce drift by 50 and 90 %, respectively, in experiments in a flower-bulb crop (Porskamp et al., 1997). Similar efficiencies of tunnel sprayers were found in orchards, vineyards and hops (Schmidt, 2001).
- The drift reduction efficiency of windbreaks (hedge- and tree rows) varies strongly with plant species and leaf stage. For regulatory risk assessment purposes, FOCUS (2004a) recommend using the following values for reduction in drift deposition for windbreaks: 25 % for bare trees, 50 % for most trees, and 90 % for full leaf stage.
- In orchards and vineyards, sensor-equipped sprayers, which prevent spraying in the gaps between plants, can reduce drift by 50 % (Koch and Weisser, 2000; Schmidt, 2001).

- Formulations and tank additives affect spray quality, and the effect of spray tank solution on droplet size is different for the different nozzle types.
- Spray drift reduction can vary with distance from the field edge. Hence, classification of a sprayer may differ from country to country although based on the same dataset.

FOCUS (2004b) presented lists of drift mitigation measures and their grouping into effectiveness classes (50-99 % drift reduction for field crops, 50-90 % for orchards) for different EU countries (Germany, UK, Netherlands, Sweden). Moreover, FOCUS (2004a) proposed to create and maintain a database of the effectiveness and applicability of spray drift reduction techniques for use in regulatory risk assessments.

The review of Ucar and Hall (2001) investigated the impact of windbreaks on pesticide drift losses. According to these authors, drift reduction offered by windbreaks apparently arises from two main causes: i) reduction in the within-crop wind speed which is responsible for droplet off-target movement, and ii) increased droplet capture within the target crop and windbreak. They reported pesticide drift reduction by 60 to 90 % due to the presence of windbreaks. Furthermore, Ucar and Hall (2001) concluded that natural (live) windbreaks are much more effective in wind speed reduction and drift mitigation than artificial ones, and that in general, medium-dense windbreaks offer an optimum porosity and thus allow for the best protection. Very dense windbreaks, in contrast, cause an undesirable wall effect, which significantly reduces the efficiency of the windbreak. However, Ucar and Hall also state that, with airflow usually being three-dimensional, compressible and turbulent, the very complex airflow near wind barriers makes it difficult to optimize windbreak design or predict their effectiveness.

In a field drift experiment in the Netherlands, De Snoo and De Wit (1998) found that a 3 m wide no-spray cropped buffer decreased drift deposition in a ditch by at least 95 %. With a 6 m wide buffer zone no drift deposition in the ditch could be measured for wind speeds not exceeding 4.5 m s^{-1}). Drift deposition in the ditch increased sharply with wind speed. For the effectiveness of no-spray buffers of varying width, official drift databases exist for Germany (Ganzelmeier et al., 1995; Rautmann et al., 2001) and the Netherlands (Van de Zande et al., 2003).

Brown et al. (2004b) observed in field trials in Canada that a 10 m wide vegetated field margin or fencerow provided adequate protection from herbicide drift into a simulated wetland area under wind conditions considered acceptable for spraying (less than 4.0 m s^{-1} wind speed). For higher wind speeds, adequate protection was afforded by the same 10 m margin plus a dense windbreak (25 % porosity) or by the margin plus a 20 m unsprayed buffer zone. Walklate (2001) observed typical drift reduction efficiencies of 86-91 % for a 7 m high alder windbreak.

Miller and Lane (1999) performed a wind tunnel experiment and found that the horizontal drift profiles from air-induction design were on average only 13.6 % of those from an equivalent design of flat fan nozzles. Ganzelmeier and Rautmann (2000) gave a brief overview of available drift-reducing sprayers and their potential effectiveness: field sprayers with injector nozzles (up to 75 % drift reduction), tunnel sprayers in vines (> 90 %), sprayers with green (foliage) detectors in vines (25-50 %), modified conventional sprayer with air assistance and injector nozzles (75 % in orchards after first tests, 90 % in hops). The authors pointed out the need for regular inspections of field sprayers.

It can be concluded that there are many possible effective measures of drift reduction and also many possibilities of combining two or more measures. While sufficient knowledge exists for suggesting default values for the efficiency of single measures, little information exists on the effect of the drift reduction efficiency of combined measures. More research on possible interactions between different drift mitigation measures and the resulting overall drift reduction efficiency is therefore required.

Finally, further drift mitigation is possible by training sprayer operators and by occasional controls of spraying practice. This would prevent improper practices such as overspray of surface water bodies and ignoring legally prescribed minimum spraying distances. For instance, Frede et al. (1998) found faulty spray practice in more than 60 % of the examined cases in a small catchment in Central Hesse, Germany.

3.4 Point sources

The effectiveness of mitigation strategies for point-source inputs of pesticides at farm scale is not too meaningful, since point source inputs from a farm can in principle be reduced to zero if the farmer follows best management practice. Hence, it is better to assess the effectiveness of mitigation measures against point sources at the catchment scale. There is limited literature available on the effectiveness of mitigation measures for point-source inputs of pesticides (Table 1), and a major part of it belongs to the “grey” literature. Jaeken and Debaer (2005) discussed a range of mitigation strategies for point sources:

- Stewardship initiatives and application of best management routines attained a reduction of total river load of 40-95 % in a range of catchment studies, e.g. Vemmenhög (Kreuger and Nilsson, 2001) or Fontaine du Theil (Maillet-Mezeray et al., 2004). However, not all stewardships were as successful. More research is needed to define critical success factors and their interaction. Organizational aspects are very important as well as the active support from various stakeholders involved.
- In-field cleaning is an effective method to reduce the amount of leftover taken back to the farmyard. Dilute spraying over the field of tank mix leftovers is not regarded as a threat to

the environment as long as the spraying takes place within the field of use and respects the registered use for the respective crop. Sprayer cleaning efficiency depends on several factors (e.g. Ramwell et al., 2004a): the time interval between spraying and cleaning, the cleaning protocol, the choice of the rinsing nozzle, the active ingredient and its formulation, and the volume of cleaning water used. Pesticide residues on the outer surfaces of the spraying equipment are predominantly located on the spray boom, the nozzles, and the spray tank. Sometimes, however, also the tractor body and the mudguards are exposed. Due to the long lifetime of spraying equipment (for sprayers often more than 15-20 years), the implementation of machinery standards for easier and more effective cleaning can take some time.

- On-farm handling is the alternative approach to in-field cleaning. All filling and cleaning operations are concentrated on a professionally equipped filling and cleaning place. The approach offers the advantage of a more conditioned environment where chemical store, water supply, personal protective equipment and first aid are at hand. The main disadvantage is the potential risk of introducing a new problem area.
- One possible approach for treatment of pesticide-contaminated water is bioremediation. The most popular bioremediation concepts are the *biobed* (Torstensson and Castillo, 1997) and other similar approaches (*Phytobac*®, *biofilter*). They basically consist of a hole in the ground (or containers) filled with a mixture of chopped straw, peat and topsoil. Biobeds are generally more robust than soils in their degrading capacity, but water loading can have a large impact on the efficiency of pesticide removal efficiency by the filter material.
- A collection system for empty containers, which was established in a case study in Belgium, yielded a collection rate of more than 90 %.
- A mandatory inspection every three years of sprayers in use was established in Belgium in 1995. This measure mitigates not only drift inputs into surface water bodies (cf. Ganzelmeier and Rautmann, 2000), but also point-source inputs due to the detection of leaks.

Ramwell et al. (2004b) investigated pesticide residues on the external surfaces of field crop sprayers and their potential environmental impact. They found that the quantity of these pesticide residues may be sufficient to be harmful to aquatic organisms if they entered a water course. Ramwell et al. (2004b) further concluded that if all residues were removed by cleaning in the field and the washings catchment area was smaller than 15 m^2, overdosing could occur, particularly for pesticides with low application rates such as pyrethroids.

Felgentreu and Bischoff (2006) found that the “recycling” of biobed leachates (i.e., re-application on the biobed) further increased pesticide removal efficiency of the biobeds, and recommended this practice for general adoption. Except diuron and isoproturon, all examined active substances were adsorbed and/or degraded in the biobed to more than 99.99 %. It should be noted in this context that biobeds and related concepts produce pesticide-contaminated waste (e.g. filter material or process water), and that this waste has to be disposed safely. This may involve significant costs depending on the country.

The effectiveness of awareness-building campaigns at the catchment scale is given by the percentage of farmers reached and convinced. Fischer et al. (1998) reported that a targeted information and advisory campaign reduced isoproturon loads in the outflows of three sewage plants in the autumn season for an intensively agriculturally used region in Central Hesse, Germany, by 50-80 %, and metazachlor loads to non-detectable levels. For a fourth sewage plant, the authors found that the effect of an advisory campaign was still observable after two years, although pesticide loads had already doubled compared with the loads measured directly after the campaign (cf. Fischer et al., 1996). Fischer et al. (1998) concluded that a single campaign is probably not sufficient to produce a permanent change in the handling of pesticides in the farmyard, and that long-term information and advice, e.g. by extension services and the industry, are desirable.

In the 9 km^2 large Vemmenhög catchment in Southern Sweden, a targeted information campaign directed to farmers was initiated in late 1994 (Kreuger and Nilsson, 2001). In the following years, different actions were taken, both on a national level (e.g. temporary economic compensations for farmers for implementing mitigation measures) and a regional level (e.g. personal visits at farms). Pesticide concentrations in the stream dropped by more than 90 %, although total applied amounts did not decrease in tendency. The authors attributed these decreasing levels of pesticides to an increased awareness among the farmers on better practices for the correct handling of spraying equipment and application procedures, including the practice of total weed killing on farmyards. However, it should be mentioned that the total herbicide glyphosate, which was increasingly used by farmers both in the fields and on farmyards, was not included in the water quality monitoring programme. Moreover, the number of farmers applying pesticides in the area had continually decreased in the 1990's, resulting in fewer possible point sources.

During an information and awareness campaign in the Nil catchment in Belgium during the years 2000 and 2001, a significant decrease of pesticide loads in the river was observed. However, when the campaign was finished in 2002, pesticide loads immediately rose again (Beernaerts et al., 2002, cited by Holvoet et al., 2005).

The Water Catchment Protection Project (The Voluntary Initiative, 2005) is part of The Voluntary Initiative http://www.voluntaryinitiative.org.uk/Content/Default.asp, a collaboration between the crop protection and farming industries and the water industry in the UK with the aim to identify practical approaches to reducing pesticide residues in water. In six pilot catchments across the UK, a continually updated “toolkit” of measures is applied and tested, containing e.g. local meetings, farm adviser visits, newsletters etc. Preliminary results and conclusions of the project are (The Voluntary Initiative, 2005):

- Although the VI project achieved good progress in most of the 6 pilot catchments since its start in 2001, one catchment (it was not stated which one) has so far failed to show any improvements in water quality.
- The awareness campaign deserves strengthening as some farmers still do not know or understand that there is a problem with surface water quality due to pesticides.
- Experience to date indicates it takes about 15 months for each catchment to start showing positive results.
- Climatic patterns in the form of wet periods leading to high drainflow can outweigh the success of the information campaign.

The Upper Cherwell catchment (199 km^2) in SE-England with predominantly clay soils is one of the pilot catchments in the Water Catchment Project, but had already been monitored for several years before. Rose et al. (2000) reported that in a 1 km^2 catchment at the headwaters of the River Cherwell, in the season 1998-1999 40 % of the total IPU load in surface water were caused by farmyard runoff. In the next season, point-source inputs were reduced by simple on-farm mitigation measures by 95%, but diffuse inputs were 10 times higher than the year before because heavy rainfalls caused drainflow events shortly after application. According to Hillier (pers. comm., 2006), the available isoproturon monitoring time series of the River Cherwell suggest that the rainfall regime in a given year is a major factor explaining IPU concentration levels in the river, and that the effect of information and awareness campaigns is hence likely to be limited. It may therefore be possible that the Cherwell catchment, due to its hydrological characteristics, is dominated by diffuse pesticide inputs at least in wet years, and the mitigation potential of point source mitigation measures might thus be very limited.

Several projects in Europe have dealt or deal with mitigation of pesticide point-source inputs into water bodies (cf. ECPA, 2003; EUREAU, 2001). The training of sprayer operators to reduce point-source inputs is the aim of the National Register of Sprayer Operators NRoSO (http://nroso.nptc.org.uk), which has recruited over 20000 active professional sprayer operators in UK. The recently launched EU-wide TOPPS project (www.topps-life.org) is aimed at identifying and disseminating advice, training and information at a larger co-ordinated scale in Europe with the intention of reducing losses of pesticides into ground - and surface water. The new “Hot Spots” project in Germany, which is funded by the German Ministry for Agriculture, aims at quantifying point-source inputs of pesticides into surface waters, back-tracking their sources, and developing suitable strategies for avoiding and/or reducing point-source inputs.

The literature examined can be summarized as follows:

- Point-source inputs can be relatively easily mitigated against by increasing awareness of the farmers with regard to pesticide handling and application, and encouraging them to implement loss-reducing measures as part of “best management practice”. Information and advisory campaigns and trainings were found to be successful and effective in most pilot catchments, but continuous effort is necessary to prevent backsliding.
- In some catchments which are dominated by diffuse inputs at least in some years, mitigation of point-source inputs alone is not sufficient to reduce pesticide loads/ concentrations in water bodies to an acceptable level.

3.5 Effect of combinations of mitigation measures at regional/catchment scale

With regard to the overall effectiveness of combinations of mitigation measures, the following can be inferred:

- If mitigation measures have efficiencies that are independent of each other, i.e. one measure does not influence the effectiveness of the other, and if mitigation measures for one pathway do not affect pesticide losses via another pathway, the effect of a combination of mitigation measures on total pesticide losses will be additive (mitigation measures for different pathways, e.g. combination of edge-of-field buffer strips and filling/cleaning operations on a biobed) or multiplicative (same pathway, e.g. combination of application rate reduction and edge-of-field buffer strips).
- However, if mitigation measures do not have efficiencies independent of each other and/or lead to increased or decreased pesticide losses via another pathway (e.g. combination of conservation tillage, subsurface drains and shift of the application date to mitigate runoff), overall loss reduction efficiencies for combinations of mitigations measures are not straightforward to obtain. In such cases, overall efficiencies have to be determined by including and simulating the mitigation measures of concern in the same model run.

4 PRACTICABILITY OF MITIGATION MEASURES AND RECOMMENDATIONS FOR IMPLEMENTATION IN PRACTICE

The literature reporting on the effectiveness of mitigation measures in reducing pesticide losses and improving water quality demonstrates that results in terms of reduction of contamination are very variable and can even be contrasting, depending on climate patterns and locations. Still, the need to put actions in place to decrease pesticide contamination requires the overall effectiveness of mitigation measures to be assessed. Table 4 provides such an assessment on the basis of the literature examined in the present review work. The table presents a list of mitigation actions discussed in section 3, grouped by input pathway. Every mitigation measure is evaluated in terms of its effectiveness and its practicability (this includes cost-effectiveness) and the assessment is used to sort measures as “recommendable” or “non-recommendable”. It should be noted that the effectiveness estimates are subjective by nature and may therefore only reflect the views of those involved in the review work.

Not only the cost-effectiveness of a mitigation measure, but also its ecological benefit and other side-effects (beneficial or detrimental) should be taken into account when deciding which mitigation measures are to be implemented in a given case.

input pathway	mitigation measure	pesticide load reduction effectiveness		practicability	recommendable for use as mitigation measure?
		at farm scale	at catchment scale	ease of implementation, further benefits, obstacles, additional costs, impact on farming systems, disadvantages, risks	
runoff/ erosion	application rate reduction	≈ percentage of rate reduction	≈ percentage of rate reduction	easy to implement, less pesticide costs, possible risk of insufficient pest/weed/disease control	yes
	shifting application to earlier or later date	potentially very high [1)] but very variable	potentially very high but variable [2)]	easy to implement, possible risk of insufficient pest/weed/disease control	yes
	buffer strips at lower field edge	variable (low to very high)	high [3)]	easy to implement, maintenance necessary, loss of arable land area for the strip and thus of crop yield	yes
	riparian buffer strips	low [4)]	very low [5)]	easy to implement, but trees grow slowly; high ecological and recreational value [4)]; possible increase of pest/disease pressure	yes
	constructed wetlands	very high (but well tested only for strongly sorbing pesticides)	medium	high installation costs, need for maintenance, installation not everywhere possible, loss of arable land area, potential problems with conservation laws [6)]	yes
	grassed waterways	high	high	easy to implement, maintenance necessary, loss of arable land area [7)] and thus of crop yield	yes
	conservation tillage	runoff: inconsistent results; erosion: probably effective, but insufficient data	unknown	easy to implement, mitigates soil erosion, in humid climates possible problem of fungal diseases → higher use of fungicides needed	yes (only for pesticide erosion losses)
	ground cover (cover crops)	insufficient data	unknown	easy to implement, mitigates soil erosion	yes (only for pesticide erosion losses)
	type of formulation	insufficient data	unknown	-	no
	subsurface drains	high	high	installation costs can be high, maintenance necessary, possible problems with pesticide losses via drainflow, in some countries regulatory disadvantage (restrictions on some pesticides for use on drained land)	depends

drainage	application rate reduction	≥ percentage of rate reduction	≥ percentage of rate reduction	easy to implement, less pesticide costs, possible risk of insufficient pest/weed/disease control	yes
	shifting application to earlier date in autumn or later date in spring	potentially very high, but very variable [8)]	potentially very high, but variable [2)8)]	easy to implement, possible risk of insufficient pest/weed/disease control	yes
	pesticide incorporation into topsoil	inconsistent results	unknown	easy to implement, suitable only for certain uses (soil herbicides/insectides/fungicides)	no
	fine topsoil tilth	low (weakly or moderately sorbed pesticides) or potentially detrimental (strongly sorbed pesticides)	low or potentially detrimental	moderately easy to implement (dependent on soil texture), possible drawback is enhanced soil erosion	no
	reduce drain efficiency	low	low	easy to implement, possible deterioration of trafficability, workability and crop growth, increase of surface runoff and associated pesticide losses	no
	collection ponds for drain outflow	unknown	unknown	-	no
leaching	application rate reduction	≥ percentage of rate reduction	≥ percentage of rate reduction	easy to implement, less pesticide costs, possible risk of insufficient pest/weed/disease control	yes
	shifting application to earlier date in autumn or later date in spring	potentially high, but variable [8)]	potentially high, but variable [8)]	easy to implement, possible risk of insufficient pest/weed/disease control	yes
	fine topsoil tilth	low or potentially detrimental	low or potentially detrimental	moderately easy to implement (dependent on soil texture), possibly enhanced soil erosion	no
drift	application rate reduction	= percentage of rate reduction	= percentage of rate reduction	easy to implement, less pesticide costs, possible risk of insufficient pest/weed/disease control	yes
	no-spray buffers	function of width, see official drift tables	function of width, see official drift tables	easy to implement, often high ecological value [9)], slight loss in yield, possible increase of weed/pest/disease pressure and thus need for higher application rates	yes

	natural windbreaks (hedges and tree rows)	low (no foliage) to very high (full leaf stage)	low to very high	easy to implement, but grow slowly; high ecological value; mitigate also wind erosion; possible increase of pest/disease pressure; possible problems with conservation laws [6)]	yes
	riparian buffer strips	low (no foliage) to very high (full leaf stage) [4)]	low to very high	easy to implement, but trees grow slowly; high ecological and recreational value [4)]; possible increase of pest/disease pressure	yes
	spray additives and formulations	low to medium	low to medium	easy to implement; very coarse drops might not grant sufficient distribution on foliage	yes
	drift-reducing nozzles (incl. air assistance)	medium to very high; see official classification on nozzle label	medium to very high; see official classification on nozzle label	easy to implement, little additional costs [10)]; very coarse drops might not grant sufficient distribution on foliage	yes
	band sprayers	very high	very high	high costs for purchase [11)]; not applicable for all crops	yes
	shielded sprayers	medium	medium	high costs for purchase [11)]; not applicable for all crops	yes
	sensor-equipped sprayers	medium	medium	high costs for purchase [11)]; only for orchards and vineyards	yes
	tunnel sprayers	very high	very high	high costs for purchase [11)]; only for orchards, vineyards and hops	yes
all diffuse sources	product substitution [12)]	zero to high	zero to high	higher or lower price, higher or lower effectiveness in weed/pest/ disease control, possible shift of risk to another pathway	depends
point sources	information campaigns	dichotomic: very low or very high	potentially high	difficulty to reach the whole farming community; campaign has to be carefully designed and conducted, continuous effort necessary	yes
	filling and cleaning operations on a biobed	very high [13)]	very high [13)]	high costs of installation and maintenance; produces toxic waste that has to be disposed safely; risk of leaching when biobeds are not state-of-the-art (compartments have to be closed at bottom and walls)	yes
	filling and cleaning operations on the field	very high	very high	high requirements for spraying equipment → often not feasible with older sprayers; possible risk of transport to surface water or of overdosing	yes
	sharing equipment or spraying by contractors	-	potentially high	easy to implement, reduces number of filling/cleaning operations, possible risk of less careful handling by contract sprayer operators	yes
	regular inspection of sprayers	-	depends on proportion of faulty sprayers	easy to implement, reduces both drift and point source inputs	yes

[1] effectiveness classes (tentative): very low = 0-20 % reduction of pesticide inputs into water bodies, low = 20-40 %, medium = 40-60 %, high = 60-80 %, very high = 80-100 %

[2] The variability in effectiveness due to different time periods between application and the next runoff/erosion or drainage event should be reduced when upscaling to the catchment scale.

[3] The variability in local effectiveness of edge-of-field buffers should cancel out when upscaling to the catchment scale.

[4] The value of riparian buffer strips arises mainly from their drift mitigation potential and their ecological functions.

[5] The catchment-scale effectiveness of riparian buffers with respect to runoff and erosion is lower for downstream-situated (level 3 and 4 streams) and higher for upstream-situated (level 1 and 2 streams) buffers (e.g. Lacas et al., 2005). The former are more common, though, in European agricultural landscapes.

[6] Constructed wetlands and windbreak hedges might be seen by authorities not as pollutant filters, but as habitats to be protected. The consequence of this would be that farmers would not install wetlands or windbreak hedges at all.

[7] However, grassed waterways are usually planned and installed so that they receive runoff not from only one, but from several fields. The loss in arable land due to the grassed waterway might thus be acceptable.

[8] However, when pesticides are applied too early in autumn on cracking soils the shifted application could have a detrimental effect.

[9] With respect to the no-spray buffers, crop-free no-spray buffers may facilitate easier enforcement of drift mitigation as suggested by FOCUS (2004a), but may be inferior to cropped no-spray buffers from a conservationist point of view. De Snoo (2001) stressed the ecological value and importance of unsprayed crop margins, especially cereal margins, for arable plant species and the associated insect fauna. He found that in winter wheat the creation of unsprayed margins is associated with only little additional costs for the farmer, whereas for sugar beet and potatoes an unsprayed crop margin is infeasible due to weed and fungal disease pressure, respectively. A better option here would be to create an unsprayed cereal or grass margin.

[10] In Germany hardly any nozzles are sold/bought nowadays that are not drift-reducing (Rautmann, pers. comm., 2006).

[11] Schmidt (2001) pointed out that new developments in drift-reducing technology are necessary, but more important are effective solutions for conventional sprayers which can be implemented with low costs in a short time.

[12] Substitution by compound with more favourable physical/chemical and/or ecotoxicological properties and/or with a lower application rate.

[13] When well managed and maintained.

Table 4. Effectiveness and practicability of mitigation measures at the farm and catchment scales.

5 IMPLICATIONS AND RECOMMENDATIONS FOR MODELLING

In this section, the mitigation measures recommended in Chapter 4 for implementing at the farm and/or catchment scale are evaluated with respect to their potential for modelling (e.g. in the FOOT tools). Furthermore, default values of pesticide load reduction efficiencies for modelling are suggested on basis of our literature review. In cases where the available literature data were not sufficient, efficiencies were derived by expert judgement. Our results are summarized in Table 5. As in the previous section, it is emphasized that the estimates are inherently subjective.

For some mitigation measures, e.g. shift of the application date or installing subsurface drains, the pesticide load reduction efficiency of this measure cannot be estimated *a priori* but has to be determined in model simulations.

input pathway	mitigation measure	ease of modelling/ implementation in tools (easy/medium/ difficult)	remarks	suggested default value for **pesticide load reduction efficiency** in the modelling (based on the examined literature and expert judgement)		How we will deal with this mitigation measure in FOOTPRINT
				farm scale	catchment scale	
runoff/ erosion	application rate reduction	easy	Freundlich sorption with exponent < 1 → loss reduction < or > rate reduction	use same percentage as the rate reduction		reduce application rate in the PRZM model input
	shifting application to earlier or later date	easy	-	has to be modelled explicitly		probabilistic PRZM modelling with varied application date
	buffer strips at lower field edge	easy	effectiveness of buffer strips depends on many factors and is highly variable -> use rather conservative values	weakly and moderately sorbed (mainly dissolved in runoff water) pesticides: 50 % (5 m), 70 % (10 m), 80 % (20 m strip width) strongly sorbed (mainly adsorbed to eroded sediment) pesticides: 60 % (5 m), 85 % (10 m), 95 % (20 m strip width)		multiply PRZM output (pesticide losses, runoff volume, eroded sediment) with suggested reduction factors; possibly use dynamic efficiencies dependent on magnitude of runoff/erosion event
	riparian buffer strips	easy	few available data	weakly and moderately sorbed: 25 % strongly sorbed: 25 %	weakly and moderately sorbed: 15 % [1)] strongly sorbed: 15 % [1)]	multiply PRZM output with reduction factors
	constructed wetlands	easy	few data for weakly and moderately sorbing pesticides	weakly and moderately sorbed: 60 % strongly sorbed: 90 %	weakly and moderately sorbed: 30 % [2)] strongly sorbed: 45 % [2)]	multiply PRZM output with reduction factors
	grassed waterways	easy	few available data	weakly and moderately sorbed: 70 % (25 m length) strongly sorbed: 90 % (25 m waterway length)		multiply PRZM output with reduction factors
	conservation tillage	difficult	runoff losses: inconsistent data; erosion losses: few available data	has to be modelled explicitly		adjust PRZM input parameters
	ground cover (cover crops)	easy	few available data	has to be modelled explicitly		adjust PRZM input parameters

	subsurface drains	difficult	-	has to be modelled explicitly	not possible to simulate drains or their effect within PRZM
drainage	application rate reduction	easy	Freundlich sorption with exponent < 1 → loss reduction > rate reduction	use same percentage as the rate reduction (conservative assumption)	reduce application rate in the MACRO model input
	shifting application to earlier or later date	easy	-	has to be modelled explicitly	MACRO modelling with varied application dates
leaching	application rate reduction	easy	Freundlich sorption with exponent < 1 → loss reduction > rate reduction	use same percentage as the rate reduction (conservative assumption)	reduce application rate in the MACRO model input
	shifting application to earlier or later date	easy	-	has to be modelled explicitly	MACRO modelling with varied application dates
drift	application rate reduction	easy	-	use same percentage as the rate reduction	reduce application rate in drift formula
	no-spray buffers	easy	efficiency is function of distance	use German or Dutch drift tables	apply reduction factor to output of drift formula
	natural windbreaks (hedges and tree rows)	easy	-	trees without foliage: 25 %, intermediate foliage: 50 %, full foliage: 90 % 3) 4)	apply reduction factor to output of drift formula
	riparian buffer strips	easy	-	trees without foliage: 25 %, intermediate foliage: 50 %, full foliage: 90 % 3) 4)	apply reduction factor to output of drift formula
	spray additives and formulations	easy	reduction efficiency depends on many factors, e.g. nozzle type	20-50 % (FOCUS, 2004b) 4)	apply user-defined reduction factor to output of drift formula
	drift-reducing nozzles (incl. air assistance)	easy	-	see official classification on nozzle label, usually 50-90 % 4)	apply reduction factor to output of drift formula
	band sprayers	easy	-	90 % 4)	apply reduction factor to output of drift formula
	shielded sprayers	easy	-	50 % 4)	apply reduction factor to output of drift formula
	sensor-equipped sprayers	easy	-	50 % 4) 5)	apply reduction factor to output of drift formula

	tunnel sprayers	easy	-	90 % [4]		apply reduction factor to output of drift formula
point sources	information campaigns	difficult	efficiency depends, apart from the quality and size of the campaign, on socio-economic factors	zero (farmer not convinced) or 100 % (convinced)	percentage of farmers reached and convinced	apply user-defined reduction factor to output of HARDSPEC model
	filling and cleaning operations on a biobed	difficult	-	100 %	percentage of farms where filling and cleaning are performed on a biobed	apply user-defined reduction factor to output of HARDSPEC model
	filling and cleaning operations on the field	difficult	-	100 %	percentage of farms where filling and cleaning are performed on the field	apply user-defined reduction factor to output of HARDSPEC model
	sharing equipment or spraying by contractors	difficult	filling and cleaning operations are possibly done with less or more care than by farmers using their own sprayers	zero or 100 % (depending on where filling and cleaning operations are performed)	percentage by which filling/cleaning operations are reduced	apply user-defined reduction factor to output of HARDSPEC model
	regular inspection of sprayers	difficult	efficiency depends on state and maintenance of equipment	zero or 100 % (depending on whether sprayer was okay or was leaking and repaired)	percentage of sprayers that were found leaking and were subsequently repaired	not considered (regular inspection should be a matter of course)

[1] The catchment-scale effectiveness of riparian buffers with respect to runoff and erosion is lower for downstream-situated (level 3 and 4 streams) and higher for upstream-situated (level 1 and 2 streams) buffers (e.g. Lacas et al., 2005). The former are more common, though, in European agricultural landscapes.

[2] Constructed wetlands cannot be installed below all fields in a catchment, but only in level terrain and close to surface water bodies. Therefore their catchment-scale effectiveness in reducing pesticide inputs into surface water bodies is lower than their edge-of-field effectiveness.

[3] adopted from FOCUS (2004a)

[4] Drift reduction of these mitigation measures in comparison with standard conditions / technology varies with distance from the sprayer. However, we neglect this for reasons of simplicity.

[5] In combination with drift-reducing nozzles, sensor-equipped sprayers can achieve a drift reduction of 75 % (Koch, pers. comm., 2006).

Table 5. Recommendations for modelling the effects of selected mitigation measures for pesticide risk assessment and management purposes.

6 SUMMARY AND CONCLUSIONS

The main conclusions of our review are summarized below. Note that there is probably much more information produced than is available in the scientific literature, as many regional monitoring and mitigation campaigns are only presented at regional conferences, but are not published (Dubus, pers. comm., 2006).

There are considerably more mitigation measures (and literature on mitigation) available for the pathways runoff/erosion and spray drift than for drainage and leaching. Of all mitigation measures, vegetated buffer strips for mitigating pesticide runoff and erosion inputs into surface water have received the largest attention in the literature.

The effectiveness of grassed buffer strips located at the lower edges of fields has been demonstrated in general. However, this effectiveness is very variable, and the variability cannot be explained by strip width alone. Riparian buffer strips are most probably much less effective than edge-of-field buffer strips in reducing pesticide runoff and erosion inputs into surface waters. Constructed wetlands are promising tools for mitigating pesticide inputs via runoff/erosion and drift into surface waters, but their effectiveness still has to be demonstrated for weakly and moderately sorbing compounds.

Pesticide runoff and drainage losses are mutually dependent. Subsurface drains are an effective mitigation measure for pesticide runoff losses from slowly permeable soils with frequent waterlogging.

Reported mitigation measures available for the pathways drainage and leaching are very limited in comparison to those available for runoff/erosion and spray drift. The effects of pesticide formulation, tillage operations and pesticide incorporation into the soil on pesticide losses via drainage and leaching are insufficiently known and at best unpredictable. These measures are therefore not suitable for recommendation as mitigation measures for pesticide losses via drainage or leaching, which leaves rate reduction, product substitution and shift of the application date as only feasible mitigation measures for both pathways. For drainage, the use of collection ponds for drain outflow analogously to constructed wetlands seems a further possible alternative, but there are no experimental data available so far on their effectiveness.

There are many possible effective measures of spray drift reduction and also many possibilities of combining two or more measures. While sufficient knowledge exists for suggesting default values for the efficiency of single measures, little information exists on the effect of the drift reduction efficiency of combined measures. More research on possible interactions between different drift mitigation measures and the resulting overall drift reduction efficiency is therefore indicated.

Point-source inputs can be mitigated against by increasing awareness of the farmers with regard to pesticide handling and application, and encouraging them to implement loss-reducing measures of “best management practice”. Information and advisory campaigns and trainings were successful and effective in most study catchments, but continuous effort is necessary to maintain farmer awareness and prevent backsliding. In catchments dominated by diffuse inputs at least in some years, mitigation of point-source inputs alone may not be sufficient to reduce pesticide loads/concentrations in water bodies to an acceptable level.

Results of the present review work will be integrated in the FOOT tools used at the local (FOOT-FS) and the catchment/regional scale (FOOT-CRS) to recommend mitigation measures to reduce pesticide contamination of water resources.

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Annex Slideshow: Selected mitigation measures.

Runoff/Erosion:

Fig. 4. Edge-of-field buffer strip (Réal et al., 2006).

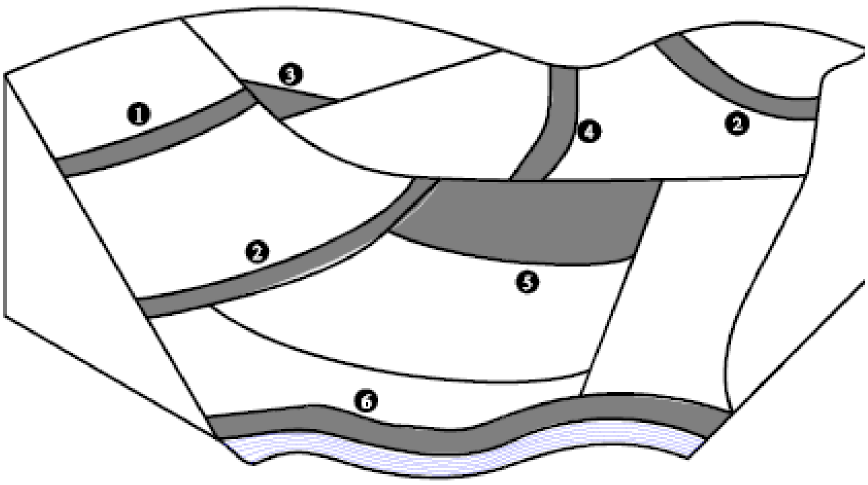

Fig. 5. Possible location of grassed buffer zones: 1) in the field, 2) at the margin of the field, 3) at the corner of the field, 4) grassed waterway,5) meadow, 6) along the riverside (FOCUS, 2004b).

Fig. 6. Grassed waterway (USDA, 2000)

Fig. 7. Contour grassed buffer strips (USDA, 2000)

Fig. 8. Vegetative barriers (USDA, 2000)

Fig. 9. Filter strips between a field and a water body (USDA, 2000)

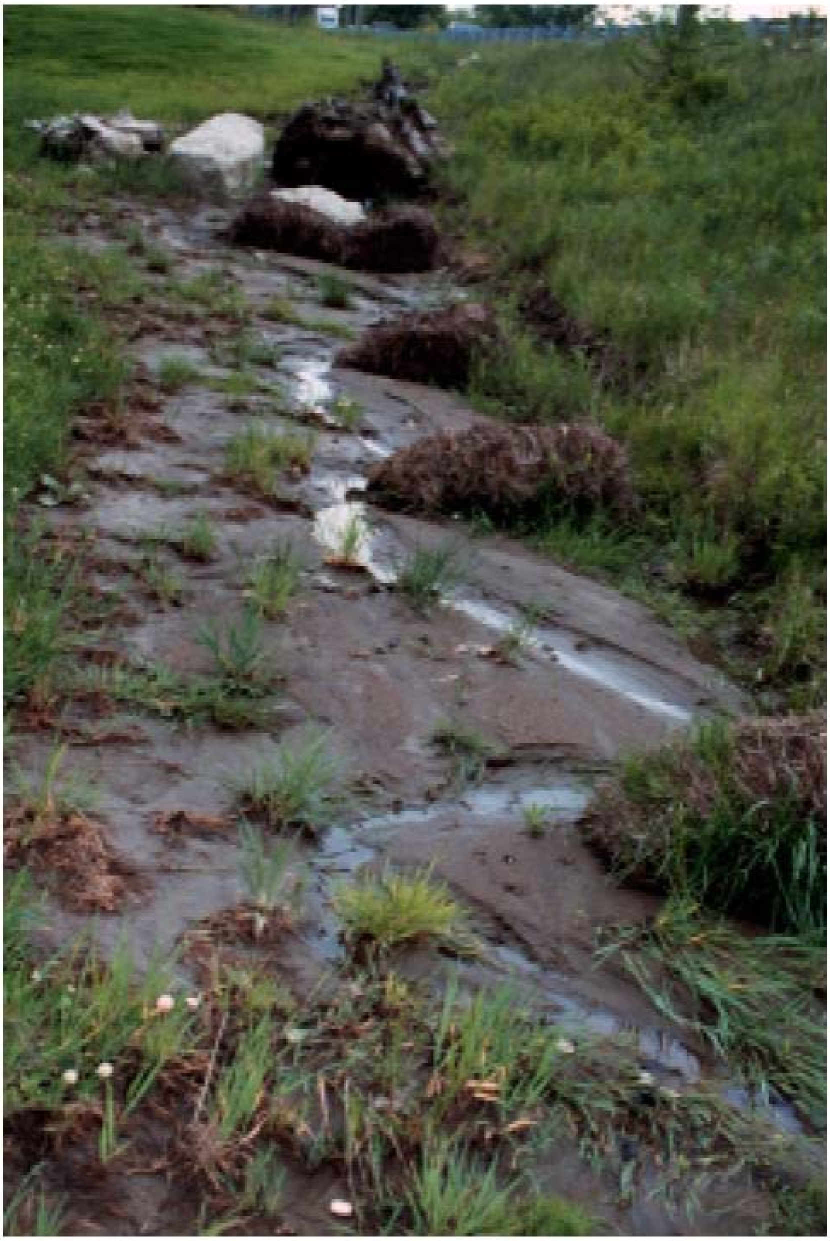

Fig. 10. Concentrated flow through a grass buffer (USDA, 2000)

Fig. 11. Conservation tillage (ARVALIS, 2006)

Fig. 12. Constructed wetland (USDA, 2000)

Spray Drift:

Fig. 13. Air induction nozzles (Meline et al., 2006)

Fig. 14. Artificial windbreak (Brown et al., 2004b)

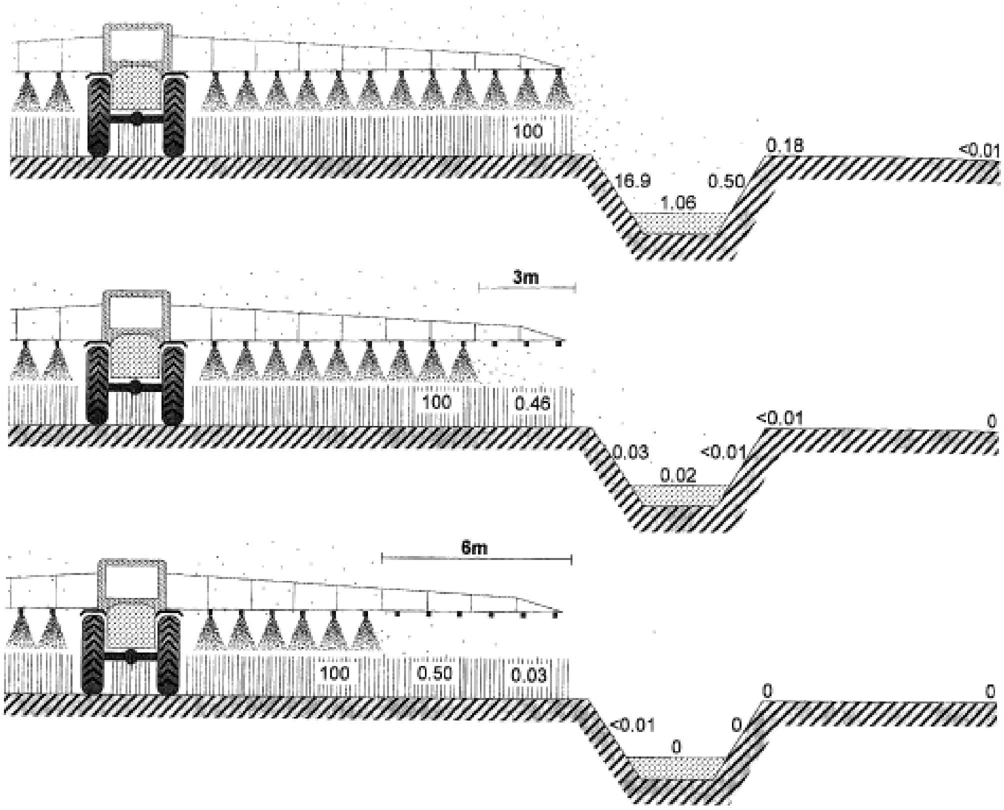

FIG. 4. Average drift deposition at a wind speed of 3–4.5 m/s at different distances from the sprayed field. (Top) Sprayed situation, (middle) unsprayed crop edge 3 m wide, (bottom) unsprayed crop edge 6 m wide.

Fig. 15. Effect of drift mitigation by leaving field margins unsprayed (De Snoo and De Wit, 1998)

Fig. 16. Riparian forest buffer (USDA, 2000)

Fig. 17. Field borders (USDA, 2000)

Fig. 18. Natural (live) windbreaks/shelterbelts (USDA, 2000)

Fig. 19. Herbaceous wind barriers (USDA, 2000)

Point sources:

Fig. 20. Upturned 'empty' containers, with rinsate leaking – a source of potential water contamination. (Higginbotham, 2001)

Fig. 21. Cat litter and other inert material can be used to absorb spills and drips. (Higginbotham, 2001)

Fig. 22. Modified biobed-filter with stacked serial filter units (Jaeken & Debaer, 2005)

Fig. 23. Different models of biobeds used in practice (Torstensson, 2000)