SWAT developments and recommendations for modelling agricultural pesticide mitigation measures in river basins

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Abstract Pesticides are useful for agriculture because of their ability to protect crops against pests. At the same time, excessive loading of pesticides in water bodies can produce toxic conditions that harm sensitive aquatic species, and render the water unfit for human consumption. Therefore, measures need to be designed, evaluated and undertaken in order to reduce pesticide pollution. In this study we focus on the Nil catchment, a small basin situated in the centre of Belgium. The necessary database and a watershed model (Soil and Water Assessment Tool—SWAT) were available to simulate different agricultural management scenarios. In order to make the model accurately predict pesticide loading to the river and instream transport, it was necessary to make several modifications to the source code. Special attention was given to implement an estimator for point losses (e.g. cleaning of spray equipment) and droplet drift, and improve the representation of physical processes in filter strips. The closing of mass balances is also described. Once the model was modified and calibrated, it could be used to simulate the pesticide mitigation strategies and evaluate their effectiveness. The simulation results revealed that strip-cropping seems to be more efficient than the sowing of cover crops, contour farming, the construction of filter strips, a 40% reduction of point losses and finally conservation agriculture. Several recommendations are given for further improvement of SWAT for management use.

Key words agricultural management practices; diffuse losses; filter strips; pesticides; point losses; river water quality modelling; surface runoff; SWAT

Développements de SWAT et recommandations pour la modélisation de mesures de réduction des pesticides agricoles dans les bassins versants

Résumé Les pesticides sont utiles en agriculture parce qu'ils protègent les cultures contre les agressions biologiques. En même temps, une concentration excessive des pesticides dans les masses d'eau peut engendrer des conditions toxiques vis-à-vis de certains organismes aquatiques et de la consommation humaine. Des mesures doivent par conséquent être définies, évaluées et mises en œuvre afin de réduire la pollution par les pesticides. Dans cette étude, nous nous concentrons sur le bassin versant du Nil, un petit bassin situé au centre de la Belgique. La base de données nécessaire et le modèle de bassin versant (Soil and Water Assessment Tool–SWAT) étaient disponibles pour simuler différents scénarios de gestion agricole. Il a été nécessaire de procéder à plusieurs modifications du code source pour obtenir des simulations précises des flux de pesticides entrant dans et transporté par la rivière. Une attention particulière a été portée à l'implémentation d'un estimateur de pollutions ponctuelles (e.g. liées au nettoyage des équipements d'épandage), l'implémentation d'une modélisation du transport de gouttelettes, et l'amélioration de la représentation des processus physiques dans les bandes tampons. La fermeture des bilans massiques est également décrite. Après modification et calage, le modèle a pu être utilisé pour simuler les stratégies de réduction de la pollution par les pesticides et en évaluer l'efficacité. Les résultats de simulation montrent que la culture en bandes semble être plus efficace que l'ensemencement de cultures de couvert, la culture en courbes de niveau, l'implantation de bandes tampons, une réduction des pollutions ponctuelles de 40% et finalement une agriculture de conservation. Plusieurs recommandations sont formulées pour une amélioration complémentaire de SWAT en tant qu'outil de gestion.

Mots clefs gestion des pratiques agricoles; pollutions diffuses; bandes tampons; pollutions ponctuelles; modélisation de la qualité de l'eau de rivière; ruissellement de surface; SWAT

1 INTRODUCTION

Agricultural water pollution is becoming a major concern, not only in developed regions such as the European Union (EU) and North America, but also in developing countries. The intensification of agricultural practices—in particular, the growing use of fertilizers and pesticides, and the specialization and concentration of crop and livestock production—have had an increasing impact on water quality. The main agricultural water pollutants are nitrates, phosphorus and pesticides (Campbell *et al.*, 2004; Clevering & Visser, 2005).

Pesticides are chemicals which are used to protect crops against pests (insects, weeds, etc.). Crop damage can reduce yields and crop quality, or even kill the crop in some cases. As a result, farmers have sought ways to reduce this damage by using pesticides. Mostly fungicides (approx. 43%), followed by herbicides (36%), insecticides (12%) and other pesticides are used in the European Union. In Belgium, almost 2.5 million tons of pesticides are applied in the field each year, atrazine and alachlor being the most widely used (Beernaerts *et al.*, 2002; EU, 2005).

Although pesticides are useful to society, a lot of problems come with their use. The most widespread and well-known issue is the occurrence of pesticides in surface water and groundwater, leading to toxicity to aquatic organisms. To ensure that risks are minimized, several measures can be taken. These pollution prevention farming methods are known as Best Management Practices (BMPs). Pesticide pollution of surface waters can be by either point losses (e.g. through leaking tools), or diffuse sources (i.e. mostly through runoff and droplet drift). By its particular nature, diffuse water pollution is more difficult to control than point losses. The pollution occurs over a wide area and its sources are difficult to identify. Further, it also varies over time and space, and depends not only on rainfall patterns and the land—slopes and soil characteristics—but also on the farmers' land management, crop choices and production techniques (Campbell *et al.*, 2004).

The simplest way to limit pesticide pollution is to reduce or eliminate their use. This can be accomplished through guided pest control (Campbell *et al.*, 2004), biological control and by means of integrated pest management (van Lenteren, 2000). If pesticides are to be used, an effective approach is to release a smaller quantity and/or less toxic pesticides into the environment while handling (i.e. reducing point losses). Second, practices should be used that minimize the transport of pesticides to surface waters, i.e. erosion control practices and drift reduction measures (Ritter & Shirmohammadi, 2001; Campbell *et al.*, 2004).

However, managing the use of pesticides needs a better understanding of the behaviour of the chemicals on the land and in the river (Müller *et al.*, 2003), its effect on the environment, and the prediction of the effectiveness of pesticide mitigation strategies. In this respect computer modelling can be a very efficient tool. Several deterministic models predict pesticide fluxes towards surface waters on a watershed scale, and are also capable of simulating BMP features. Information about such models can be found in the reviews of Hantush & Kalin (2003) and Borah & Bera (2003). Some describe long-term effects of hydrological and water management practices, such as AnnAGNPS (Annualized Agricultural Non-Point Source model) (Bosch *et al.*, 1998), HSPF (Hydrological Simulation Program FORTRAN) (Donigian *et al.*, 1993) and SWAT (Soil and Water Assessment Tool) (Arnold *et al.*, 1998). As far as we know, none of them has implemented relationships describing point losses and droplet drift. For our study, the SWAT model was selected because of the existence of an extensive manual, the GIS interface, and the freely available open-source code, allowing the user to perform modifications easily. Furthermore, this model is increasingly being used to predict pesticide fluxes to rivers (Kannan *et al.*, 2007), and is put forward in the reviews from Borah & Bera (2003) and Quilbe *et al.* (2006).

The objective of this work is to evaluate the effectiveness of different agricultural management practices in reducing pesticide fluxes towards the river, by using a modelling approach. With this objective in mind, modifications had to be done to the source code to make the model accurately predict pesticide loading to the river and the instream processes. Finally, several recommendations are given for further improvement of SWAT for management use.

2 CASE STUDY

The Nil catchment is a small and hilly basin situated in the central part of Belgium, southeast of Brussels. It drains an area of 32 km^2, 7% of it being inhabited, and the main crops grown are

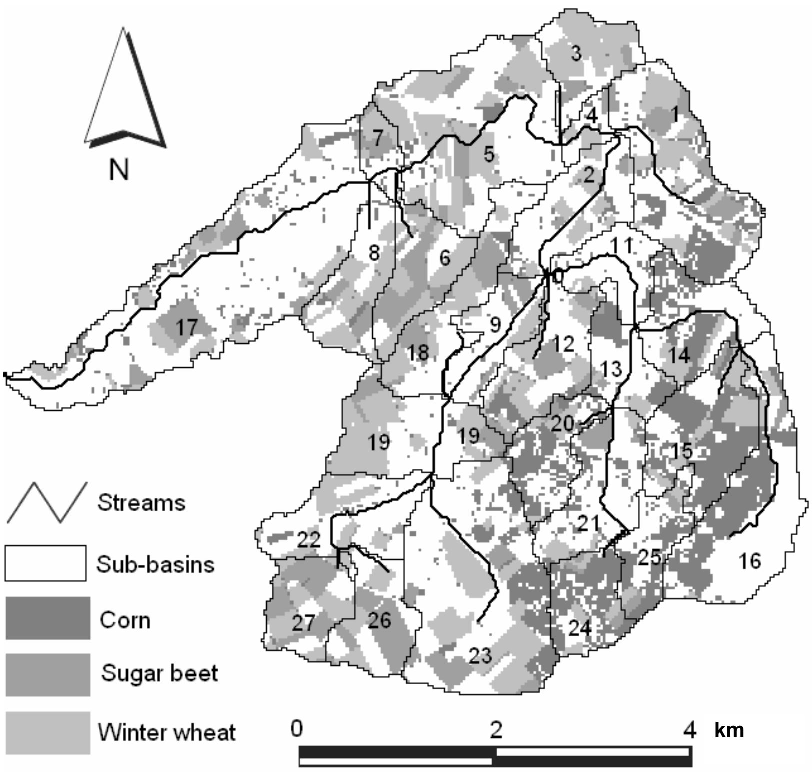

Fig. 1 Land use (1999) in the Nil catchment (adapted from Romanowicz *et al.*, 2003). The sub-basins and streams are drawn onto it.

winter wheat (22% of the catchment area), corn (15%) and sugar beet (10%) (Fig. 1). The river itself is 14 km long and has a retention time of about 1 day. The predominant soil type is loam. There are no drainpipes and no wastewater treatment plants situated in the catchment. The basin is characterized by a low baseflow (124 mm water on average annual basis), which results from its specific geological structure. Highly permeable Brusselian sands lie above a less permeable rock (Abdeslam, 1998). Hereby, on average 357 mm water is routed annually to the deep aquifer and, hence, contributes to flow in the adjacent River Train.

The Nil catchment was selected because it is a well documented basin, studied in detail in terms of pesticide application, by the Center for Research in Animal Health and Agro-chemistry (CODA) (Beernaerts *et al.*, 2002). They performed inquiries during the spring periods (e.g. the time of pesticide application) of 1998–2002, in which they asked farmers to give detailed information concerning the amount of pesticide applied, the application dates, the kinds of pesticides utilized for different crops and the treated surface. Forty-two percent of the farmers provided this information (CODA, 2003). This extensive data set was available for the present work.

The use of pesticides by farmers in the Nil basin leads unavoidably to water pollution of the river. To gain insight into the occurrence of these pollutants in the different compartments of the river, CODA performed a monitoring campaign during the same period (CODA, 2003). Two years later, the Flemish Institute for Technological Research (VITO) performed more intensive measurements in the Nil. Detailed information about the two monitoring campaigns can be found in Beernaerts *et al.* (2002) and Holvoet *et al.* (2007). Several pesticides were measured in these two campaigns, of which the herbicide atrazine was chosen as the study object. High concentrations of this pesticide, applied on corn fields in the Nil catchment, could be found after rainfall events, due to runoff, and even more significant concentrations were observed in dry weather conditions, due to droplet drift and point losses on application days.

3 MODELLING TOOL, SWAT

The SWAT model was developed by the US Department of Agriculture, Agricultural Research Service (USDA-ARS) (Arnold *et al.*, 1998) to predict the impact of land management practices on water, sediment and amounts of chemicals originating from agriculture, in large complex river basins with varying soils, land use and management conditions over a long period of time. It is a partly physically-based and partly distributed, continuous model with a daily calculation time step. As such, the model is not designed to simulate detailed, single-event flood routing.

Due to the intrinsic spatial-temporal variability of watersheds, a geographical information system (GIS) is an essential and efficient method for collecting, storing and retrieving input data required for simulation models like SWAT. Integration of the model in the ArcView version 3.0a software, along with the Spatial Analyst version 1.1 extension, resulted in the user-friendly tool AVSWAT. For our study, we used the AVSWAT2000 version of the graphical user interface for SWAT (Di Luzio *et al.*, 2002). The SWAT ArcView system consists of three main components: (1) a pre-processor generating sub-basin topographic parameters and model input parameters; (2) an editor for the input data set and a simulation executor; and (3) a post-processor to view graphical and tabular results.

The basic map inputs needed by the comprehensive model are specific for each case study, and include a gridded digital elevation model (DEM), soil map, land-use/cover map, hydrographical map (stream lines) and climate information. In addition, the interface allows land-use designation, soil, weather, groundwater, water use, management, soil chemistry, pond and stream water quality data to be entered, as well as the simulation period, to ensure a successful simulation. The collection and calculation of input data for the Nil catchment is described in detail in Holvoet *et al*. (2005). Special attention was given to use of high-resolution maps. In this case, maps with a spatial resolution of 30 m were added to the model interface.

Within SWAT, a catchment is divided into "hydrologically" connected sub-basins based on the topographic features of the watershed, and, further, each sub-basin is split into hydrological response units (HRUs)—lumped areas in a sub-basin consisting of unique combinations of land use, soil type and management (Neitsch *et al.*, 2002b). In our study, the Nil basin was divided into 27 sub-basins and further into 227 HRUs.

No matter what type of problem is studied with SWAT, the water balance is the driving force behind everything that happens in the watershed. Simulation of the hydrology of a watershed can be separated into two major parts.

- The first part is the land phase of the hydrological cycle. It controls the amount of pesticide loading to the main channel in each sub-basin. The model simulates its movement into the stream network via surface runoff (in solution and bound to sediment transported by the runoff), and into the soil profile and aquifer by percolation (in solution). The equations used to model the movement of pesticide in the land phase of the hydrological cycle were adopted from the Groundwater Loading Effects on Agricultural Management Systems (GLEAMS) (Leonard *et al.*, 1987). Pesticide transport by water and sediment is calculated for each runoff event, and pesticide leaching is estimated for each soil layer when percolation occurs. SWAT partitions groundwater into two aquifer systems: a shallow, unconfined aquifer and a deep, confined aquifer. The shallow aquifer contributes baseflow to streams within the watershed. Water and chemicals entering the deep aquifer are modelled as a mass flow lost from the system (Arnold *et al.*, 1993). In this study, this approach is justified as, in reality, the deep groundwater of the Nil catchment is lost to the adjacent River Train.
- The second part is the water or routing phase, which can be defined as the transport of pesticides through the channel network of the watershed to the outlet. The model divides the total pesticide load in the channel into dissolved and sediment-attached components. Pesticide transformations in the dissolved and sorbed phases are governed by first-order decay relationships. The major in-stream processes simulated by the model are settling, burial, resuspension, volatilization, diffusion and transformation (Biesbrouck *et al.*, 2002; Neitsch *et al.*, 2002b).

4 MODIFICATIONS OF THE SOURCE CODE

4.1 Original source code

To make reliable predictions of pesticide concentrations in a river, it is important to pay special attention to the simulation of hydrology. A calibration of the most sensitive hydrological parameters was performed for the period 1997–2000, followed by a validation for the period 2001–2004, as described in Holvoet *et al.* (2005). For the period 1997–2004, the Nash-Sutcliffe coefficient (Nash *et al.*, 1970) improved from a value of –25.7 to +0.32 (Holvoet *et al.*, 2005), which is in agreement with values found in the literature (Gassman *et al.*, 2007; Moriasi *et al.*, 2007).

In a next step, pesticide characteristics and management were added to the model. By comparing model results with measurements, some hiatus and inaccuracies in the source code, more specifically in the module for river routing, were highlighted. Furthermore, the original SWAT model cannot represent the observed pesticide peaks during dry periods (i.e. peaks originating from point losses and drift), a clear model deficiency (Holvoet *et al.*, 2008). With the objective to simulate and compare different agricultural management practices, modifications were made to get a model that accurately predicts the pesticide loading to the river system and the instream processes.

4.2 Model improvements for predicting loadings to the river

Different steps were taken to simulate the loading to the river system. Holvoet *et al.* (2005) made a first attempt to differentiate between direct (i.e. point losses and drift) and runoff losses. In this study, further distinction is made between point pollution and droplet drift. Furthermore, the SWAT model is improved to better represent the physical processes occurring in a vegetated filter strip.

4.2.1 Implementing point losses Point losses are caused by the cleaning of spray equipment on paved surfaces, the leakage of tools, spills, etc. (Gerecke *et al.*, 2002). They can lead to very high peak concentrations due to the absence of the extra dilution caused by rainwater runoff, causing large acute effects on the ecosystem, especially in headwaters where the dilution is small. In order to model point-source pollution, the meaning of the application efficiency parameter (ap_{ef}) in the model was changed. Originally, this parameter refers to the fraction of the pesticide application rate that is not lost from the system, and will reach the foliage or the soil surface. In the adapted code, the application efficiency coefficient expresses the fraction of the initial pesticide dose that is not diverted directly to the river as point losses, and will be applied on the field through spray equipment. The improvement to the source code existed in a modification in the module for pesticide application, which can be written as:

$$\text{point\_loss} = \text{dose} \times \left(1 - \text{ap}_{\text{ef}}\right) \times \text{area\_HRU} \times 10^8 \quad (1)$$

where point_loss is the amount of pesticide diverted directly to the river (mg); dose is the initial amount of pesticide (kg ha^{-1}); ap_{ef} is the application efficiency; area_HRU is the HRU area on which the pesticide is applied (km^2); and 10^8 is a unit conversion factor. For each application day, the model calculates the amount of initial dose lost via point-source pollution. These losses are summed up with the outputs of the land-phase section (e.g. runoff losses) in the SWAT model, and are thus directly diverted to the river.

4.2.2 Implementing an estimator for droplet drift Pesticide drift is the transport of pesticide droplets at the time of pesticide application, or soon thereafter, from the field to any non-target site, including surface waters (Hewitt, 2000). A distinction can be made between droplet and vapour drift. Droplet drift is defined as liquid droplets formed by spray nozzles that are carried out of the treated field by wind and are deposited (De Schampheleire *et al.*, 2007). Vapour drift includes the evaporation of droplets from plants or soil surfaces, and the transport of the resulting active ingredient followed by the deposition on any non-target site (Carlsen *et al.*, 2006). The latter is rather small in contribution (due to the low vapour pressure of atrazine), so the focus is put on pesticide losses by droplet drift. The calculation of these loadings was based on the German drift database (Ganzelmeier *et al.*, 1995), in accordance with the recommendation of the FOrum for the Co-ordinaten of pesticide fate models and their USe (FOCUS) Surface Water workgroup. The data in these tables were generated from a series of studies (at a number of locations—minimum 1 m distance from the latest crop row—and with a variety of crops). For arable crops a simple regression equation was obtained:

$$\%\text{drift} = A \times z^{B} \tag{2}$$

where %drift is the percentile drift value at distance z from the latest crop row (m); and A and B are the constant and exponential regression factor, respectively, dependent on the crop type and the growth stage (FOCUS, 2001). As the focus in this work is on the application of atrazine on corn fields, values for arable crops with three applications were looked for in the German tables. The values found for A and B, are 2.0244 and –0.9956, respectively (Ganzelmeier *et al.*, 1995). Note that equation (2) is only valid when there is at least one metre distance between the latest crop row and the edge of the treated field (and thus the water body). This is also the minimum distance imposed by the Belgian Federal Government when spraying arable crops (Huyghebaert *et al.*, 2005). The German database is useful for pesticides that are applied in compliance with the principles of good agricultural practice (GAP). This comprises the application during low wind velocities, the use of approved equipment, as well as the application under favourable climatic conditions. For our case study, it can be reasonably assumed that those conditions are fulfilled. Further, the Ganzelmeier drift data are 90th percentile worst-case drift values obtained for wind directions perpendicular to the receiving water bodies and at wind velocities at the upper end of the conditions compliant with GAP. As such, they have to be considered as conservative.

In order to calculate the total drift loading on a receiving water body, equation (2) must be integrated over the width of the receiving water body, resulting in equation (3). Therefore, factor B should be larger than –1.

$$\overline{\text{Drift}\%} = \frac{A}{(z_2 - z_1) \times (B+1)} \times \left(z_2^{B+1} - z_1^{B+1}\right) \tag{3}$$

where $\overline{\text{Drift}\%}$ is the mean drift percentage (%), z_2 and z_1 are the distances (m) from the latest crop row to the farthest and closest edges of the water body, respectively. To determine the drift loading into the surface water (mg) on each application day, the mean drift percentage needs to be multiplied by the area of the receiving water body (m^2) and the dose applied on the field thus, after correction for the loss by point discharges (kg ha^{-1}). This leads to equation (4):

$$\text{drift\_loss} = \text{dose} \times \text{ap}_{\text{ef}} \times \overline{\text{Drift}\%} \times \text{area\_water} \tag{4}$$

The area of the receiving water body (m^2) can be calculated as follows:

$$\text{area\_water} = \sqrt{\text{area\_HRU}} \times W \times 1000 \tag{5}$$

where W is the channel width (m) and 1000 is a unit conversion factor. The length of the channel along an HRU was set equal to the square root of the HRU area (km^2), thus assuming that an HRU has the form of a square. As in SWAT, the HRUs are lumped areas and no information about the position of the original fields is available. Therefore, as it is not known whether a field borders the stream according to its length or its width, the assumption that HRUs have a square shape is an intermediate solution.

When a filter strip is placed along the water course, a smaller fraction of the applied pesticide dose will enter the river system by drift. The drift losses in the river are, like the point losses, summed to the outputs of the land phase section. The effective amount of pesticide applied on the field (reaching the foliage or the soil surface) can then be calculated by subtracting point losses, drift losses to the river and the amount of drifted pesticide that is captured in the filter strip, from the initial application dose.

In cases where data concerning wind direction, wind velocity, air temperature and humidity, as well as nozzle type, spraying pressure and tractor driving speed are missing, the Ganzelmeier approach forms an acceptable estimator for drift.

4.2.3 Improving the representation of physical processes in filter strips Vegetated filter strips form a physical and biochemical barrier between the water course and the source of pollution, i.e. the agricultural field. The purpose of these zones (sown with permanent vegetation like grass) is to trap sediment, plant nutrients, organic matter and chemicals such as pesticides, as runoff from cropland passes through the vegetated area. Several processes are responsible for this function, i.e. filtration, infiltration, absorption, adsorption, uptake, volatilisation and deposition, with the predominant processes tending to be infiltration of dissolved pollutants and deposition of sediment-attached pollutants (Gharabaghi *et al.*, 2000; Brown *et al.*, 2004; Lacas *et al.*, 2005).

In order to calculate the mass of sediment, nutrients and pesticides that is trapped by a filter strip, the original SWAT model uses a conservative filter strip trapping efficiency. In the original code this variable is calculated as follows (Neitsch *et al.*, 2002b):

$$\text{trap}_{\text{ef}} = 0.367 \times (\text{width}_{\text{filtstrip}})^{0.2967} \quad (6)$$

where trap_{ef} is the fraction of the constituent loading trapped by the filter strip, and $\text{width}_{\text{filtstrip}}$ is the width of the filter strip (m). This formula has the following limitations:

(a) The maximum width of the filter strip is 30 m. Wider strips result in a trapping efficiency that equals one. In reality, this will not be valid under all circumstances and under all rainfall intensities.

(b) The model does not take into account hydrological variations in runoff scenarios. In reality, the trap efficiency will be different for storm events and normal rainfall events, i.e. the rain intensity will play a role.

(c) For each compound, a similar fraction is retained in the filter strip. In reality, there will be a difference between coarse and small particles, and between dissolved and bound fractions. The coarser particles will sediment faster and pesticides, for example, are mainly bound to the small clay particles.

(d) The model considers a filter strip for a whole hydrological response unit (HRU) and not only for the fields truly situated along the river.

(e) The formula was derived from US empirical data on filter strip efficiency (Bärlund *et al.*, 2007).

All the above limitations and inaccuracies render it impossible to make valid predictions about the efficiency of the filter strip. Therefore, the source code was modified by extending the processes in the infiltration part and by adding a part describing sedimentation in the filter strip. The extension of the infiltration part consists of the addition of runoff from the adjacent field to the rainfall in the filter strip. This amount of water (i.e. rainfall and runoff) can then infiltrate together with the dissolved nutrients and pesticides. Hereby, it is assumed that infiltration of sediment particles in the soil can be neglected. Secondly, sedimentation of particles from the runoff water was not incorporated into the model and had to be implemented in the source code. Different models could be used to describe this process. The most cited mathematical model is the Kentucky model (Tollner *et al.*, 1976), which was successfully validated for certain field conditions. However, the model is far from accurate in assessing the sedimentation of small particles, to which pesticides and other pollutants are mainly attached. A better method for the assessment of sediment deposition was proposed by Deletic (2000). Based on her laboratory studies at the University of Aberdeen, UK, a correlation between the trapping efficiency for the sediment fraction s (particles of diameter d_s) $T_{r,s}$ and the particle fall number $N_{f,s}$ was obtained (Deletic, 2000, 2001):

$$T_{r,s} = \frac{N_{f,s}^{0.69}}{N_{f,s}^{0.69} + 4.95} \quad (7)$$

The particle fall number $N_{f,s}$ can be expressed as follows:

$$N = \frac{b \times v_s}{h \times v_m} \tag{8}$$

where b is the width of the grass strip (m), v_s is the Stokes' settling velocity of the particle d_s (m s^{-1}), h is the depth of flow (m), and v_m is the mean flow velocity between the grass blades (m s^{-1}). The velocities were defined as:

$$v_s = \frac{g}{18 \times \eta}(\delta_s - \delta_w) \times d_s^{\,2} \tag{9}$$

$$v_m = \frac{q^{0.4} \times \text{slp}^{0.3}}{n^{0.6}} \tag{10}$$

where η is the dynamic viscosity of water (kg m^{-1} s^{-1}); δ_w and δ_s are the density of water and the sediment particle (kg m^{-3}), respectively; d_s is the mean particle diameter (m); q is the overland flow rate per unit width (m^2 s^{-1}); slp is the average slope of the filter strip (m m^{-1}); and n is the Manning's roughness coefficient of the strip (s m$^{-1/3}$). The slope was assumed to be 0.1%, and the overland flow rate per unit width (q) could be calculated by dividing the peak runoff rate Q_{peak} (m^3 s^{-1}) by the length of the filter strip b (m). The peak runoff is the maximum runoff flow rate that can occur for a given rainfall event, and is calculated by the SWAT model with a modified rational formula (Neitsch *et al.*, 2002b). Using the maximum runoff flow rate instead of the daily runoff flow rate leads to a better prediction of the real efficiency of the filter strip in catching the sediment particles. The particles were divided into three fractions, namely clay, loam and sand, where the distribution in the runoff flow was assumed to be the same as in the soil. In order to estimate the depth of flow (h) in equation (8), the calibrated Manning's equation was used (Tollner *et al.*, 1977):

$$h = \frac{1.5 \times q \times n}{R_r^{2/3} \times S_{\text{veg}}^{1/2}} \tag{11}$$

The unknown parameters in equation (11) are the hydraulic radius R_r (m) and the vegetation spacing S_{veg} (m). The latter was assumed to be 4.5 mm (Gharabaghi *et al.*, 2000), while the former can be computed as follows (Tollner *et al.*, 1977):

$$R_r = \frac{S_{\text{veg}} \times h}{2 \times h + S_{\text{veg}}} \tag{12}$$

The following assumptions underlie the above equations: (a) particles already accumulated in the grass do not resuspend, and (b) the grass is never submerged. After determination of the trap efficiency of each sediment fraction, the mass of each component (i.e. nutrients, sediment particles and pesticides) trapped by the filter strip is calculated.

4.3 Model improvements for river routing

With the extension of the source code for point-source pollution, relatively large amounts of pesticides may enter the river during low flow periods, especially in small catchments like the Nil. Under these conditions, some inaccuracies in the mass balances were revealed by van Griensven *et al.* (2006). First, the threshold for minimum flow that activates water quality calculations was set too high in the original code. SWAT only calculates chemical dynamics when the limit value for flow (0.01 m^3 s^{-1}) is exceeded. Second, numerical errors appear in the river routing, which has a major impact on the pollutant mass balance during low flow conditions with significant loads. Hence, several modifications had to be done to the source code, such as reducing the threshold for water quality calculations to 0.000001 m^3 s^{-1}, and implementing a new routing module in the code.

Originally, water was routed through the channel network using the Muskingum method or the variable storage method. In the modified code, a new routing module was implemented in which the relation between velocity and flow is calculated by solving the Manning equations in an iterative way. Further corrections were done to the routing component for river bed infiltration and river evaporation. The effect of those modifications is not visible in the flow performance, but is represented in changes of the hydrological state variables (flow velocity and depth of flow). The latter affect the pesticide outputs both in total export and in shape, due to the loss of soluble pesticide with the infiltrating or evaporating water (van Griensven *et al.*, 2006a).

5 MODELLING AGRICULTURAL MANAGEMENT PRACTICES

After the source code was improved, a calibration of the model for pesticide supply was performed for 1998, followed by a validation for the period 1999–2002, as presented in Holvoet *et al.* (2008). The modified model could then be used to compare different measures for pesticide reduction.

In Fig. 2, the contribution of droplet drift, point and runoff losses to the total load of dissolved atrazine is represented for the mass coming from sub-basin 25 (with land use consisting of 37.12% corn) towards reach 25 (with length of 64 m) during the application period of atrazine in 1998. As can be seen from this figure, the input of dissolved atrazine by droplet drift is very low compared to point and runoff losses. Measures that limit point losses and runoff flow will thus be of interest in view of reducing atrazine fluxes towards the Nil. Given their small contribution, drift mitigation strategies (e.g. drift reducing nozzles) will not be discussed further, with the exception of the filter strips that also have an effect on it.

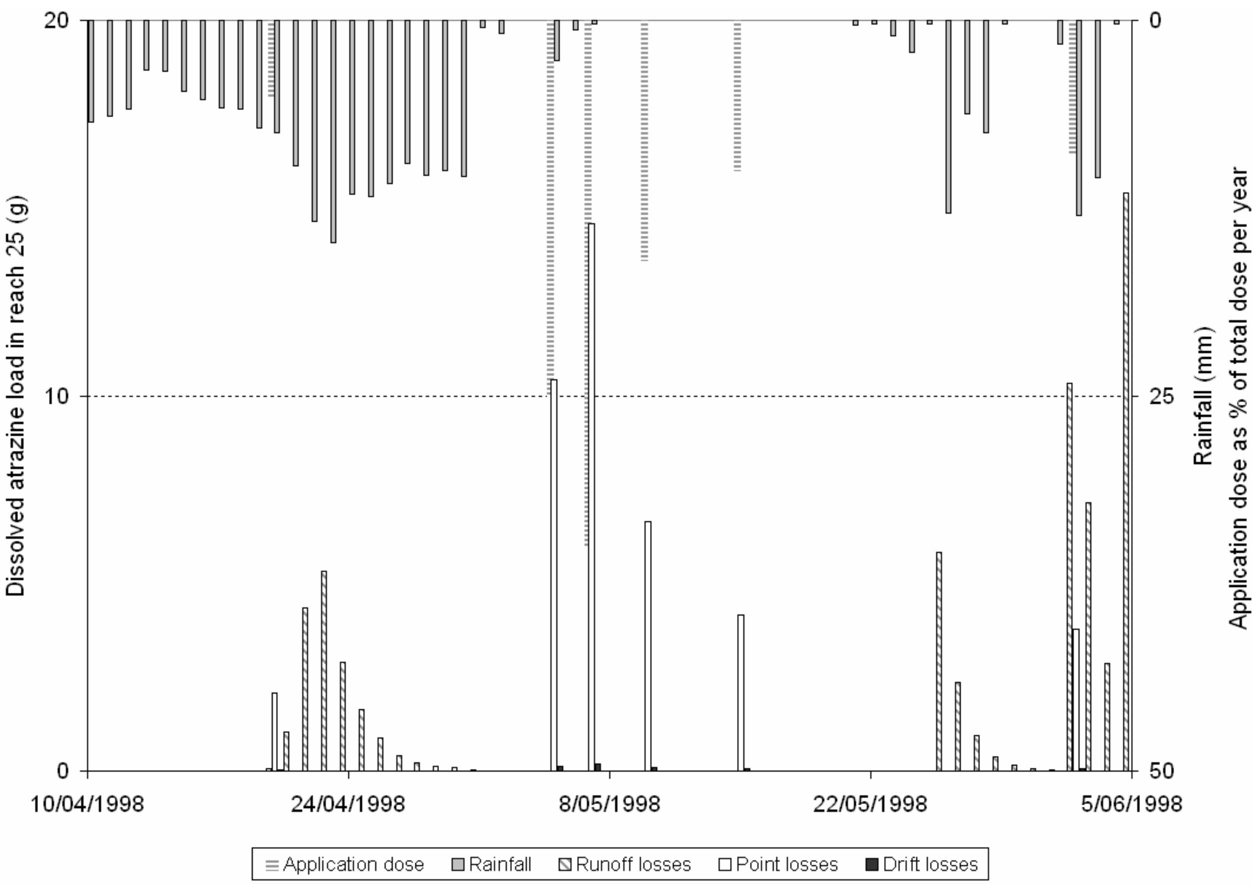

Fig. 2 Predicted load of dissolved atrazine coming from sub-basin 25 during spring 1998, together with the measured rainfall and the initial pesticide dose (showed as hanging bars on the secondary axis).

Table 1 Simulated load of atrazine in the river Nil for different values of the application efficiency parameter, together with the percentage increase or decrease (year 1998).

Application efficiency parameter (ap_{ef})	0.998	0.9985	0.9991	0.9997
Load of atrazine in solution	2.58	2.38	2.12	1.87
Increase(+)/decrease(–) in load (%)	8.80	0	–10.59	–21.14

5.1 Measures to reduce point pollution

Point losses can be reduced by careful pesticide handling e.g. avoiding spills, leaking tools, etc. Next to them, a simple prevention technique is avoiding spray rests in the spray tank. When these cannot be avoided, they should be removed with treatment options like a biobed, biofilter or phytobac. Despite the ease of applying these mitigation measures, point-source pollution can contribute up to 50–70% of the load of pesticides found in Belgian rivers like the Nil. Based on the study of Beernaerts *et al.* (2002), it seems that permanent awareness raising among the farmers remains necessary.

A reduction of point losses can be simulated by setting the application efficiency parameter (ap_{ef}) in the model to a higher value. The calibrated value of this parameter for the Nil case was 0.9985 (Holvoet, 2006). Results of the simulations for the year 1998 for different values of the application efficiency parameter are presented in Table 1. A reduction of the point losses by 80% results in a decrease of the atrazine load by 21%; while a reduction by 40% leads to almost 11% of load reduction. It should be noticed that at this stage no conclusion could be drawn on the most efficient point-source pollution prevention technique.

5.2 Measures to reduce diffuse pollution

Several agricultural management practices can result in a reduction of runoff losses to the river. Within this study, and keeping in mind the landscape characteristics of the Nil catchment, five erosion control practices were considered: conservation tillage, sowing cover crops, contour farming, strip-cropping and construction of filter strips (Holvoet, 2006). One can remark that the latter management technique will also reduce drift losses. More information on those practices can be found in Evans *et al.* (2003) and Ritter & Shirmohammadi (2001).

Conservation agriculture was simulated by changing the ploughing practice in the management file of the SWAT model. Minimum tillage (i.e. using a chisel plough instead of a disk plough, or only preparing a seedbed) and no-till management (i.e. direct seeding) are two forms of this agricultural management practice. It should be noted that a mouldboard plough is not used in conservation agriculture. To simulate the sowing of cover crops (in our study rye), contour farming and strip-cropping, the model parameters Moisture Condition II Curve Number (CN2) and/or support practice factor (USLE-P) were modified. CN2 is an empirical parameter in the SCS curve number equation used for estimating surface runoff at the average moisture content of the soil. The model parameter USLE-P is defined as the ratio of soil loss with a specific support practice to the corresponding loss with up-and-down slope culture, and is part of the modified Universal Soil Loss Equation (Neitsch *et al.*, 2002b). The different scenarios for the five practices are represented in Table 2. For filter strips the following steps need to be performed before one can simulate this practice with the modified model:

(a) All fields/HRUs that have a filter strip should be identified in the catchment. This was established by manipulating the land-use map using GIS software, whereby a new land-use class was created for areas sown with corn near the river (COWA). Also, for the filter strips a new land-use class was defined, termed COBU. For the implementation case, Bermuda grass was assumed to be the vegetation type in these strips. Its long roots combined with relatively limited canopy height provide a good buffering capacity. The characteristics of this grass are given in the model manual (Neitsch *et al.*, 2002a).

(b) It is essential that both the fields-with-filter-strips and the filter strips themselves are presented as HRUs. The watershed is divided into HRUs based on a user-defined threshold for land use classes and soil classes (both set to 10%). Land uses that cover a percentage of the sub-basin area below the threshold will be eliminated. The second scale further excludes marginal soil classes within the land-use area. Since the land-use classes that represent fields-with-filter-strips and the filter strips themselves, in particular, cover small areas, those classes would be excluded in this step. In this study, their representation was guaranteed by setting the threshold for land-use classes to zero, resulting in a large number of HRUs at the expense of an increase in simulation time. For that reason, we reduced the number of land-use classes to four: CORN (corn fields not foreseen as a filter strip), COWA, COBU and a general class.

(c) A link between the HRUs representing the field-with-filter-strip and the filter strip itself, respectively, was made in the *.hru file of the field-with-filter-strip by inserting the number of the filter-strip-HRU. Hereby, it is necessary for the number of the filter-strip-HRU to be larger than that of the field-with-filter-strip.

Due to this impractical way of working, the performance of constructing filter strips along the river was checked by making an exercise for two adjacent fields situated in Sub-basin 4 (with land use consisting of 3.82% corn): one corn field-with-filter-strip and the grassed strip itself. The herbicide atrazine, applied on the corn field near the river (with a length of 657 m), can only enter the filter strip with runoff water and by droplet drift. The results of the exercise were then extrapolated for all corn fields in the catchment situated along the river (15.3%), assuming that each corn field provides the same trapping efficiency. Extrapolation is justified in this case, because the predominant soil type in all fields is loam, and there is little variation in terms of typography between the fields.

The simulation results for the studied erosion control practices are presented in Table 3 for the year 1998. Before drawing any conclusions, it should be stressed that the changes of the CN2 and

Table 2 Model parameters/management inputs used to represent the different BMPs.

BMP		Changed parameters/management inputs
Conservation agriculture	Mouldboard plough	Instead of disk plough
	Chisel plough	Instead of disk plough
	Only seedbed preparation	No ploughing before winter
	Direct seeding	No till management
Sowing cover crops: rye	Mouldboard plough	Instead of disk plough; and CN2: –4
	Chisel plough	Instead of disk plough; and CN2: –4
	Only seedbed preparation	No ploughing before winter; and CN2: –4
	Direct seeding	No till management; and CN2: –4
Contour farming		USLE–P: 0.6; CN2: –3
Strip-cropping		USLE–P: 0.3; CN2: –5
Vegetated filter strips	5 m width	see detailed description

Table 3 Results of BMPs for the reduction of the atrazine load (%) in the river Nil for 1998.

BMP		% increase (+) / decrease (–) in atrazine load:		
		Dissolved	Bound	Total
Conservation agriculture	Mouldboard plough	–0.15	19.90	0.96
	Chisel plough	0.33	–23.26	–0.98
	Only seedbed preparation	0.57	–32.78	–1.28
	Direct seeding	1.02	–43.84	–1.47
Sowing cover crops: rye	Mouldboard plough	–32.12	–33.97	–32.23
	Chisel plough	–32.12	–47.16	–32.96
	Only seedbed preparation	–32.08	–51.51	–33.16
	Direct seeding	–32.04	–58.43	–33.51
Contour farming		–25.93	–56.50	–27.63
Strip-cropping		–37.27	–80.79	–39.69
Vegetated filter strips	5 m width	–11.64	–12.07	–11.67

USLE-P parameter values are based on the literature and the model manual (Neitsch *et al.*, 2002a). Sensitivity analysis revealed that the model results are rather sensitive to the curve number value (Holvoet *et al.*, 2005). As a consequence, the study performed should be considered as a guide to rank the measures in effectiveness rather than as a quantitative assessment. To achieve the latter, field data would be necessary in order to be able to better parameterize the model for BMPs.

From Table 3 it is clear that strip-cropping is the most successful practice for atrazine reduction in the Nil basin, for both the dissolved and the bound atrazine fractions. The alteration of low and high erosion-resistant crops not only reduces the overland flow rate, but also decreases the transport of sediment particles and reduces the maximum slope length. Strip-cropping is followed by sowing cover crops and contour farming. The latter has the greatest impact on the attached atrazine fraction. The construction of filter strips results in a relatively limited decrease in pesticide load. It should be noted that this measure was not applied on all corn fields in the catchment, as was done with the other measures, but only on those situated along the river. Moreover, the model does not take into account runoff water coming from fields situated above. This can result in underpredicting the amount of runoff water entering the filter strip, and in underestimating the overland flow velocity. Also, the filtering capacity is only calculated if a real filter strip (i.e. a field sown with Bermuda grass) is present along the water course. Modification of ploughing practices seems the least efficient measure in this study. Ploughing with a mouldboard plough even leads to an increase in the total (and bound) atrazine load in the river. This type of plough turns the soil around in such a manner that erosion sensitive soil is brought to the top layer.

6 MODEL LIMITATIONS AND RECOMMENDATIONS

Modelling is a virtual exercise. Therefore, one should perform small-scale experiments in order to better understand pesticide transport, the impact of mitigation measures, and to evaluate the simulations of the model for Best Management Practices (BMP), including the presented modifications. For the moment, some BMP-related processes within SWAT are still described in a very simplified manner, and many of the equations use ill-defined parameters.

Another problem is that no field-scale application of agricultural management practices can be modelled since the fields are lumped together in HRUs. For instance, filter strips will only affect the runoff of those fields that border the river. Another issue is that for all fields within one HRU, the same application date and rate for pesticides and nutrients, and also the use of the same management techniques, is assumed. However, in reality, spatial variability exists due to variability in farmers and in farmers' customs. Due to the shown high sensitivity of the application date as model input (Holvoet *et al.*, 2005), it would be useful to describe this input factor as a probability distribution, in order to reflect the uncertainty related to it. In addition, a split-HRU tool in the interface could help to stochastically represent the differences between HRUs in a watershed. A step further is a truly distributed modelling approach, representing small-scale landscape elements such as individual fields and filter strips, and the relations between them. For SWAT modelling, this would primarily require development of the GIS-based pre-processor. The latter developments will only lead to improvements if such models can be fed by field-scale databases containing information on the agricultural practices with respect to time (application rate, application times, etc.) and the applied mitigation measures (filter strips, contour farming, etc.). It takes large efforts to create such databases but they currently already exist for several regions in the world.

Further, an uncertainty analysis would be helpful to evaluate and improve the model predictions, and eventually to reduce the uncertainties in the future. For this case study, Holvoet (2006) performed a preliminary analysis of the uncertainties in model parameters and inputs, showing indeed the important impact of both uncertainties on the model results.

7 CONCLUSIONS

The original SWAT model is not able to represent pesticide peaks observed in the River Nil during dry weather days (i.e. point losses and drift); only runoff-related peaks can be predicted. It was therefore necessary to modify the source code for point losses and to implement an estimator for droplet drift. The predicted values of point losses and droplet drift are rough estimates, as currently the application efficiency parameter has a constant value during simulation even though it can vary on a daily basis. Nevertheless, the suggested approximation is more correct than ignoring point losses and droplet drift. Further, a closer look at the river routing module by van Griensven *et al.* (2006a) revealed that some miscalculations existed in the original code that have a clear impact if there are significant pollutant loads transported during low-flow periods. In this study, modifications were performed to set the correct mass balances for flow and pollutants, including pesticides, over the entire simulation period. Finally, the existing variability in trapping efficiency of vegetated filter strips, both in time and in space, is represented by including in the source code physically based equations describing the infiltration and sedimentation processes in the strip. The equations used in our approach take into account the variation in soil properties, vegetation and slope of the filter strips, and the hydrological characteristics of the runoff events. Furthermore, a partitioning was made between the soluble and bound incoming compounds. As such, the results are expected to be more realistic than using the original SWAT equation that only considers the width of the filter strip. Despite the improvements performed, the practical evaluation of this measure remains hampered by the necessity to create input files with the current GIS interface.

With the modified SWAT model, different mitigation measures could be simulated and evaluated quantitatively by ranking them according to their effectiveness to diminish point and runoff losses to the water body. The results revealed that strip-cropping seems to be more efficient than the sowing of cover crops, contour farming, the construction of filter strips, a 40% reduction of point losses and finally conservation agriculture.

The developments done in this study should be seen as steps taken in the direction of better pesticide modelling at catchment scale. They should be evaluated further by means of field experiments combined with an uncertainty analysis of the model results. The proposed physically-based concepts for filter strips and a proper representation of agricultural management also require a more distributed alternative to the HRU concept of SWAT. Hereby, individual fields and their relative position in the landscape could be identified, described and modelled as separate elements within SWAT.

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REFERENCES

Abdeslam, A. (2004) Simulation numérique des écoulements souterrains et des relations nappes-rivières dans le bassin versant du Train (Belgique). PhD Dissertation, Catholic University Louvain-la-Neuve, Louvain-la-Neuve, Belgium (in French).

Arnold, J. G., Allen, P. M. & Bernhardt, G. (1993) A comprehensive surface–groundwater flow model. *J. Hydrol.* **142**(1–4), 47–69.

Arnold, M. S., Srinivasan, R. S. & Williams, J. R. (1998) Large-area hydrologic modelling and assessment: Part I. *J. Am. Water Resour. Assoc.* **34**(1), 73–89.

Bärlund, I., Kirkkala, T., Malve, O. & Kämäri, J. (2007) Assessing SWAT model performance in the evaluation of management actions for the implementation of the Water Framework Directive in a Finnish catchment. *Environ. Model. Software* **22**(5), 719–724.

Beernaerts, S., Debongie, P., De Vleeschouwer, C. & Pussemier, L. (2002) Het pilootproject voor het Nilbekken. Onderzoek naar de belangrijkste oorzaken van vervuiling van het oppervlaktewater door gewasbeschermingsmiddelen en middelen om deze te verminderen. In: *Groenboek*, Chapter 6, 33–38. Belgaqua-Phytofar, Belgium (in Dutch).

Biesbrouck, B., Wyseure, G., Van Orschoven, J. & Feyen, J. (2002) AVSWAT2000. Course, Catholic University of Leuven, Laboratory for Soil and Water Management (LSWM), Leuven, Belgium.

Borah, D. K. & Bera, M. (2003) Watershed-scale hydrologic and non-point-source pollution models: review of mathematical bases. *Trans. Am. Soc. Agric. Engrs* **46**(6), 1553–1566.

Bosch, D. D., Theurer, F. D., Bingner, R. L., Felton, G. & Chaubey, I. (1998) Evaluation of the AnnAGNPS water quality model. In: *Am. Soc. Agric. Engrs Annual International Meeting* (12–16 July 1998, Orlando, Florida).

Brown, L. C., Leeds, R., Sulc, M. R. & Van Lieshout, L. (2004) Vegetative filter strips: application, installation and maintenance. Available at: http://ohioline.osu.edu/aex-fact/0467.html.

Campbell, N., D'Arcy, B., Frost, A., Novotny, V. & Sansom, A. (2004) *Diffuse Pollution: An Introduction to the Problems and Solutions* (first edn). IWA Publishing, London, UK.

Carlsen, S. C. K., Spliid, N. H. & Svensmark, B. (2006) Drift of 10 herbicides after tractor spray application, 1. Secondary drift (evaporation). *Chemosphere* **64**(5), 787–794.

Clevering, O. A. & Visser, A. J. (2005) Beheeradvies grasbufferstroken voor het project Actief Randenbeheer Brabant. Praktijkonderzoek Plant & Omgeving BV. Tech. Report Sector AGV, Wageningen, The Netherlands (in Dutch).

CODA (2003) Available at: http://www.var.fgov.be/pdf/nil_donnees_publiques.pdf.

Deletic, A. (2000) Sediment behaviour in overland flow over grassed areas. MSc Thesis, University of Aberdeen, Aberdeen, UK.

Deletic, A. (2001) Modelling of water and sediment transport over grassed areas. *J. Hydrol.* **248**(3–4), 168–182.

De Schampheleire, M., Spanoghe, P., Brusselman, E. & Sonck, S. (2007) Risk assessment of spray drift damage in Belgium. *Crop Prot.* **26**(4), 602–611.

Di Luzio, M., Srinivasan, R. & Arnold, J. G. (2002) Integration of watershed tools and SWAT model into BASINS. *J. Am. Water Resour. Assoc.* **38**(4), 1127–1141.

Donigian, A. S., Imhoff, J. C., Bricknell, B. R. & Kittle, J. L. (1993) Application guide for Hydrological Simulation Program FORTRAN (HSPF) Environmental Research Laboratory. Report EPA/600/R-97/080, US Environmental Protection Agency, Athens, Georgia, USA.

EU (European Union) (2005) Towards a thematic strategy on the sustainable use of pesticides. Summaries of legislation. Available at: http://europa.eu/scadplus/leg/en/lvb/l21288.htm.

Evans, B. M., Lehning, D.W., Borisova, T., Corradini, K. J. & Sheeder, A. S. (2003) PRedICT Users Guide for the Pollutant Reduction Impact Comparison Tool. Penn State Institutes of the Environment, the Pennsylvania State University, Pennsylvania, USA.

FOCUS (2001) FOCUS Surface Water Scenarios in the EU Evaluation Process under 91/414/EEC. FOCUS Working Group on Surface Water Scenarios. Report, EC Document Reference SANCO/4802/2001-rev.2.

Ganzelmeier, H., Rautmann, D., Spangenberg, R., Streloke, M., Herrmann, M., Wenzelburger, H.-J. & Walter, H.-F. (1995) Untersuchungen zur Abtrift von Pflanzenschutzmitteln. Mitteilungen aus der Biologischen Bundesanstalt für Land- und Forstwirtschaft Berlin-Dahlem (in German).

Gassman, P. W., Reyes, M. R., Green, C. H. & Arnold, J. G. (2007) The Soil and Water Assessment Tool: Historical development, applications, and future research directions. *Trans. Am. Soc. Agric. Biol. Engrs* **50**(4), 1211–1250.

Gerecke, A. C., Schärer, M., Singer, H. P., Müller, S. R., Schwarzenbach, R. P., Sägesser, M., Ochsenbein, U. & Popow, G. (2002) Sources of pesticides in surface waters in Switzerland: Pesticide load through wastewater treatment plants—current situation and reduction potential. *Chemosphere* **48**(3), 207–315.

Gharabaghi, B., Rudra, R. P., Whiteley, H. R. & Dickinson, W. T. (2000) Sediment-removal efficiency of vegetative filter strips. Annual Research Report, ASAE Sacramento Convention Center, Sacramento, California, USA, 32–40.

Hantush, M. M. & Kalin, L. (2003) Evaluation of sediment transport models and comparative application of two watershed models. Tech. Report, Environmental Protection Agency, National Risk Management Research Laboratory, USA.

Hewitt, A. J. (2000) Spray drift: impact of requirements to protect the environment. *Crop Prot.* **19**(8–10), 623–627.

Holvoet, K., van Griensven, A., Seuntjens, P. & Vanrolleghem, P. A. (2005) Sensitivity analysis for hydrology and pesticide supply towards the river in SWAT. *Phys. Chem. Earth* **30**(8–10), 518–526.

Holvoet, K. (2006) Monitoring and modelling the dynamic fate and behaviour of pesticides in river systems at catchment scale. PhD Dissertation, Ghent University, Ghent, Belgium. Available at: http://biomath.ugent.be/biomath/publications/-download/.

Holvoet, K., Seuntjens, P., Mannaerts, R., De Schepper, V. & Vanrolleghem, P. A. (2007) The water-sediment as a highly dynamic system: Results of an intensive pesticide monitoring campaign. *Water Sci. Tech.* **55**(3), 177–182.

Holvoet, K., van Griensven, A., Gevaert, V., Seuntjens, P. & Vanrolleghem, P. A. (2008) Modifications to the SWAT code for modelling direct pesticide losses. *Environ. Model. Software* **23**(1), 72–81.

Huyghebaert, B., Jaeken, P. & Nuyttens, D. (2005) Maatregelen ter beperking van verontreiniging van oppervlaktewater door gewasbeschermingsmiddelen. Tech. Report, Federale Overheidsdienst Volksgezondheid, Veiligheid van de Voedselketen en Leefmilieu, Belgium (in Dutch).

Kannan, N., White, S. M. & Whelan, M. J. (2007) Predicting diffuse-source transfers of surfactants to surface waters using SWAT. *Chemosphere* **66**(7), 1336–1345.

Lacas, J.-G., Voltz, M., Gouy, V., Carluer, N. & Gril, J.-J. (2005) Using grassed strips to limit pesticide transfer to surface water: a review. *Agron. Sustain. Dev.* **25**(2), 253–266.

Leonard, R. A., Knisel, W. G. & Still, D. A. (1987) GLEAMS: Groundwater loading effects on agricultural management systems. *Trans. Am. Soc. Agric. Engrs* **30**(5), 1403–1428.

Moriasi, D. N., Arnold, J. G., Van Liew, M. W., Bingner, R. L., Harmel, R. D. & Veith, T. L. (2007) Model evaluation guidelines for systematic quantification of accuracy in watershed simulations. *Trans. Am. Soc. Agric. Biol. Engrs* **50**(3), 885–900.

Müller, K., Deurer, M., Hartmann, H., Bach, M., Spiteller, M. & Frede, H.-G. (2003) Hydrological characterisation of pesticide loads using hydrograph separation at different scales in a German catchment. *J. Hydrol.* **273**(1–4), 1–17.

Muñoz-Carpena, R., Parsons, J. E. & Gilliam, J. W. (1999) Modeling hydrology and sediment transport in vegetative filter strips. *J. Hydrol.* **214**(1–4), 111–129.

Nash, J. E. & Sutcliffe, J. V. (1970) River flow forecasting through conceptual models: Part 1. A discussion of principles. *J. Hydrol*. **10**(3), 282–290.

Neitsch, S. L., Arnold, J. G., Kiniry, J. R., Srinivasan, R. & Williams, J. R. (2002a) *Soil and Water Assessment Tool. User's Manual, Version 2000*. Agricultural Research Service and Blackland Research Centre, Texas A&M University, USA.

Neitsch, S. L., Arnold, J. G., Kiniry, J. R., Williams, J. R. & King, K. W. (2002b) *Soil and Water Assessment Tool. Theoretical Documentation, Version 2000*. Agricultural Research Service and Blackland Research Centre, Texas A&M University, USA.

Quilbe, R., Rousseau, A. N., Lafrance, P., Leclerc, J. & Amrani, M. (2006) Selecting a pesticide fate model at the watershed scale using a multi-criteria analysis. *Water Qual. Res. J. Canada* **41**(3), 283–295.

Ritter, W. & Shirmohammadi, A. (2001) *Agricultural Nonpoint Source Pollution: Watershed Management and Hydrology*. CRC Press, Boca Raton, Florida, USA.

Romanowicz, A. A., Vanclooster, M., Rounsevell, M. & La Jeunesse, I. (2003) Sensitivity of the SWAT model to the soil and land use data parameterisation: A case study in Thyle catchment, Belgium. *Ecol. Model.* **187**(1), 27–39.

Tollner, E. W., Barfield, B. J., Haan, C. T. & Kao, T. Y. (1976) Suspended sediment filtration capacity of simulated vegetation. *Trans. Am. Soc. Agric. Engrs* **19**(4), 678–682.

Tollner, E. W., Barfield, B. J., Vachirakornwatana, C. & Haan, C. T. (1977) Sediment deposition patterns in simulated grass filters. *Trans. Am. Soc. Agric. Engrs* **20**(5), 940–944.

van Griensven, A., Holvoet, K. & Yang, J. (2006a) Open sources model codes: critical review on the routing modules of the Soil and Water Assessment Tool (SWAT). In: *Proceedings of the Seventh International Conference on Hydroinformatics* (4–8 September 2006, Nice, France).

van Griensven, A. & Meixner, T. (2006b) Methods to quantify and identify the sources of uncertainty for river basin water quality models. *Water Sci. Tech.* **53**(1), 51–59.

van Lenteren, J. C. (2000) A greenhouse without pesticides: fact or fantasy? *Crop Prot.* **19**(6), 375–384.

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