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Environmental Modelling & Software 23 (2008) 72–81

Environmental Modelling & Software

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Modifications to the SWAT code for modelling direct pesticide losses

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Received 10 May 2006; received in revised form 5 January 2007; accepted 3 May 2007
Available online 29 June 2007

Abstract

In different river catchments in Europe, pesticide concentrations in surface waters frequently exceed the standards, possibly resulting in negative impacts on aquatic fauna and flora. Pesticides can enter river systems both immediately after application, i.e. as a direct loss, or with some time delay due to runoff or leaching. We define a direct loss as the sum of point losses and drift losses on an application day that will reach the river immediately after or during application. Point losses are due to the clean-up of spray equipment, leaking tools, waste water treatment plants etc. Different studies demonstrated the importance of direct losses. In small river systems, their contribution accounts for 30 to 90% of the pesticide load to surface water.

As many studies and models only partly take into account these direct losses or even not at all, we attempted to model the dynamic occurrence of pesticides also coming from these sources. For this purpose, some modifications and extensions to the SWAT (Soil and Water Assessment Tool) model were made. Special attention was paid to closing mass balances and implementing an estimator for total direct losses, drift and point losses. To verify the modifications we focused on the use of the herbicide atrazine in the Nil, a small and hilly river basin in the centre of Belgium. The modified SWAT code resulted in a better correspondence between measured and simulated atrazine concentrations and loads, in particular for direct losses. For the year 1998, the Nash–Sutcliffe coefficient improved from a value of −2.63 to 0.66. In addition, the modelling results of the test case revealed that the contribution of drift losses to the total pesticide load in the river system is rather small: even without a ‘non spray zone’, they account for only 1% of the total load. Point sources, on the other hand, contribute for 22% up to 70% of the pesticide load and need to be considered in pesticide pollution management.

The resulting model needs further testing for other pesticides and other catchments. In future, the model can be used for comparison of different measures that can be taken to minimise pesticide fluxes towards river systems and in performing realistic risk assessments.

Keywords: Atrazine; Direct losses; Modelling pesticides; SWAT

Software availability

Name of software: SWAT2000 (Soil and Water Assessment Tool)-FORTRAN.
Developer: USDA Agricultural Research Service (USDA-ARS).
Contact address: 808 East Blackland Road, Temple, TX 76502-6712, USA. Tel. +1 (254) 770-6502; Fax +1 (254) 770-6561; http://www.brc.tamus.edu/swat/
Year first available: 2000.
Hardware required: PC.
Software required: Arc View 3.2 for the AVSWAT GIS interface.
Program language: FORTRAN.
Program size: 2.1 MB (compiled executable).
Availability and cost: free download at http://www.brc.tamus.edu/swat/. The modules associated with the

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doi:10.1016/j.envsoft.2007.05.002

proposed revision may be requested to the corresponding author.

1. Introduction

Pesticides are useful to society thanks to their ability to increase crop yields by destroying disease-causing organisms and controlling insects, weeds and other pests. At the same time, most pesticides may be harmful to humans, animals and the environment because of their ecotoxicity, their potential bio-accumulating properties and their hormone disrupting effects (Cuppen et al., 2000; Van den Brink et al., 2000; Hanson et al., 2002; Yamaguchi et al., 2003; Wendt-Rasch et al., 2004; Capkin et al., 2006).

Through different pathways, pesticides can enter surface water systems. Different studies in European river basins demonstrate that the presence of pesticides in surface waters is not only due to diffuse losses (i.e. carried through runoff water, wash out to groundwater and atmospheric drift), but that point losses make an active contribution (Bach et al., 2001; Beernaerts et al., 2002; Gerecke et al., 2002; Neumann et al., 2002; Leu et al., 2004a). An intensive monitoring campaign in the river Nil in Belgium during the years 1998–2002 revealed high herbicide concentrations at the mouth of the river, even during dry and wind-still days. Consequently, these high peaks could only originate from direct losses at the day of application, i.e. drift or point losses through the clean-up of spray equipment, leaking tools, etc. This was further confirmed in the authors' own monitoring campaign in which the dynamics of the water-sediment system were followed intensively during spring 2004 (Holvoet et al., 2007). The contribution of the point losses can be decreased from 40% up to 60% by sensitizing farmers as was proven during the years 2000–2001. When this sensitization campaign was stopped in 2002, pesticide loads in the river immediately increased varying from 40% up to 80% depending on pesticide (Beernaerts et al., 2002). During the monitoring campaign of 2004 (Holvoet et al., 2007), the contribution of direct losses accounted for 60% up to 90%, although the total amount of rainfall during the campaign was smaller than in all other studied years. This proves the importance of direct losses. As the Nil was also studied in detail for pesticide applications and management practices during the period 1998–2002, we chose it as a test case for analysing the processes underlying the direct and diffuse sources through modelling and monitoring.

Some deterministic models exist that describe long-term effects of hydrological changes and water management practices on a watershed scale, such as AnnAGNPS (Bingner and Theurer, 2001), HSPF (Donigian et al., 1993) and SWAT (Arnold et al., 1996). As far as we know, none of them implemented the aforementioned direct losses. Information about the different models was found in the model manuals and in the review articles of Shoemaker et al. (2005), Borah and Bera (2003) and Cox (2003). The SWAT model was selected because of its suitability for larger catchments, the freely available open source code, the existence of an extensive manual and the GIS interface.

To account for direct losses of pesticides to surface water adaptations to the original SWAT model were needed. In a first step, the original SWAT model results were compared to the results of the intensive monitoring campaign performed during the years 1998–2002. Hereby, some hiatus and inaccuracies in the source code were highlighted. With the modelling objective in mind, adaptations were done on the source codes in order to get a model that adequately describes the diffuse and point source pollution while respecting the mass balance.

2. Case study

2.1. Catchment area

The Nil basin is a small rural, hilly basin situated in the central part of Belgium, Southeast of the capital Brussels. The average elevation amounts to 151 m a.s.l., with the highest top reaching 167 m a.s.l. and the watershed outlet lying at 110 m a.s.l. (Fig. 1). The Nil catchment drains an area of 32 km^2, is 14 km long and has a surface water retention time of about 1 day. Seven percent of the area is inhabited and the main crops grown are winter wheat (22% of the catchment area), corn (15%) and sugar beet (10%) (Fig. 2a). Eighteen percent of the catchment consists of pasture. The predominant soil type is loam. There are no drainpipes and no waste water treatment plants in the catchment.

Further, the catchment is characterised by a low baseflow which results from its specific geological structure. Highly permeable Brusselian sands, showing hydraulic conductivities between 10^{-3} and 10^{-5} m/s, lay above a less permeable socle (Abdeslam, 1998). Hereby, an important part of the groundwater of the Nil-catchment is drained to the adjacent river Train.

2.2. The Soil and Water Assessment Tool (SWAT)

The SWAT model (Arnold et al., 1998) is chosen for modelling catchment-scale pesticide fluxes to the river. It is a well-documented model with open source code, able to manage hydrology, sediments, nutrients and pesticides (Neitsch et al.,

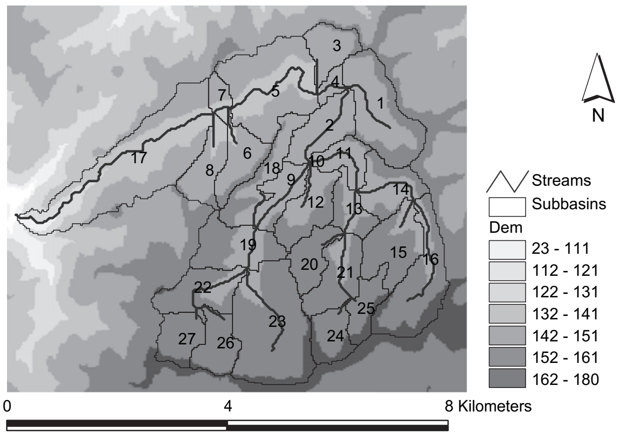

Fig. 1. Sub-basin delineation of the Nil catchment automated by means of a DEM (m).

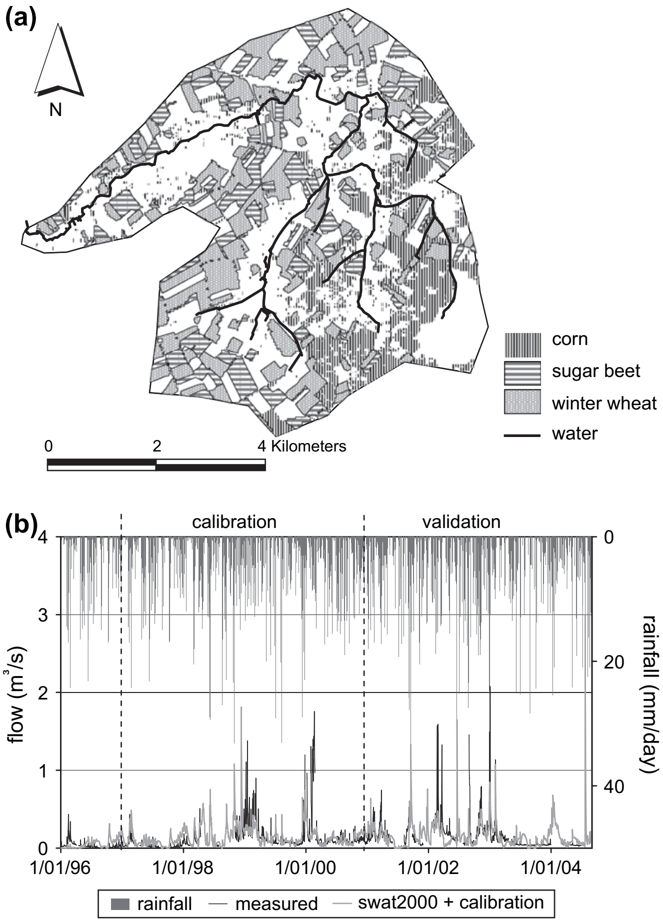

Fig. 2. (a) Land use (1999) in the Nil catchment (adapted from: Romanowicz et al., 2003). (b) Comparison of measured (black line) and simulated (grey line) flow data (1996–2000).

2002b). It is a conceptual and semi-distributed, continuous model with a daily calculation time step. Once optimised and calibrated, it can be used for optimising agricultural management (Bracmort et al., 2006; Behera and Panda, 2006; Santhi et al., 2006; Bärlund et al., 2007).

The water quantity processes simulated by SWAT include precipitation, evapotranspiration, surface run-off, lateral subsurface flow, ground water flow and river flow. The equations used to model the movement of pesticide in the land phase of the hydrologic cycle were adopted from GLEAMS (Leonard et al., 1987). SWAT simulates pesticide losses in surface run-off, sediments and percolation below the root zone. The movement of the pesticide is controlled by its solubility, degradation half-life both on plant foliage and in the soil, and soil organic carbon adsorption coefficient (Neitsch et al., 2002a). Once SWAT determines the loadings of water, sediment, nutrients and pesticides to the main channel, the loadings are routed through the stream network of the watershed using a command structure similar to that of HYMO (Williams and Hann, 1973). In addition to keeping track of mass flow in the channel, SWAT models the transformation of chemicals in the stream and streambed (Neitsch et al., 2002a).

As water flows downstream, a portion may be lost due to different processes described below. The water storage in the reach at the end of the time step is calculated:

$$V_{stored,2}=V_{stored,1}+V_{in}-V_{out}-tloss-E_{ch}+diversions+V_{bnk} \quad (1)$$

where $V_{stored,2}$ is the volume of water in the reach at the end of the time step, $V_{stored,1}$ is the volume of water in the reach at the beginning of the time step, V_{in} is the volume of water flowing into the reach during the time step, V_{out} is the volume of water flowing out of the reach during the time step, *tloss* is the volume of water lost from the reach via transmission through the bed, E_{ch} is the evaporation from the reach for the day, *diversions* is the volume of water added to the reach by rainfall or point source discharges or removed from the reach for agricultural or human use, and V_{bnk} is the volume of water added to the reach via return flow from bank storage. All units are m^3 H_2O. The outflow V_{out} can be calculated either by the Muskingum or the Variable Storage method.

During periods when a stream receives no groundwater contributions, it is possible for water to be lost from the channel via transmission through the side and bottom of the channel. Transmission losses are estimated with the equation:

$$tloss = K_{ch} \cdot TT \cdot P_{ch} \cdot L_{ch} \quad (2)$$

where *tloss* are the channel transmission losses (m^3 H_2O), K_{ch} is the effective hydraulic conductivity of the channel alluvium (mm/h), TT is the flow travel time (hr), P_{ch} is the wetted perimeter (m), and L_{ch} is the channel length (km). Transmission losses from the main channel are assumed to enter bank storage or the deep aquifer. Travel time is computed by dividing the volume of water in the channel by the flow rate:

$$TT = V_{stored}/q_{out} \cdot 3600 \quad (3)$$

where TT is the travel time (hr), V_{stored} is the storage volume (m^3 H_2O) and q_{out} is the discharge rate (m^3/s).

Evaporation losses from the reach are calculated:

$$E_{ch} = coef_{ev} \cdot E_o \cdot L_{ch} \cdot W \cdot TT/24 \quad (4)$$

where E_{ch} is the evaporation from the reach for the day (m^3 H_2O), $coef_{ev}$ is an evaporation coefficient, E_o is potential evaporation (mm H_2O), L_{ch} is the channel length (km), W is the channel width at water level (m), and TT is the travelling time (h).

2.3. Model set-up

We used the AVSWAT2000 version of the model, where the simulator is integrated in a GIS by an ArcView pre-processor (Di Luzio et al., 2002). It uses gridded DEM data, polygon/grid coverages of soils and land use, and point coverages of weather stations as basic input to the model.

Within SWAT, a catchment is partitioned into a number of sub-basins (Fig. 1), based on the threshold area which defines the minimum drainage area required to form the origin of a stream. Within the sub-basins, hydrologic response units

(HRUs) are defined, which are lumped land areas consisting of unique combinations of land cover, soil and management (Neitsch et al., 2002a).

The collection and calculation of input data, i.e. weather data, a DEM, a land use and soil map, was described in Holvoet et al. (2005).

For the simulation, the Nil was divided into 27 sub-basins and reaches. The sub-basins are further divided into 227 HRUs, as defined by land use and soil type.

2.4. Modelling hydrology

As described in Holvoet et al. (2005), by means of an LH-OAT sensitivity analysis (van Griensven and Meixner, 2006) the most sensitive parameters for hydrology could be determined. An automatic calibration of these parameters resulted in a quite good fit. Both the calibration (1997–2000) and the validation period (2001–2004) are presented in Fig. 2b. In years with higher rainfall, the model describes the data well. In dry periods, the model has some difficulties in yielding good predictions. This is due to the very low baseflow caused by the specific geological structure of the catchment (Holvoet et al., 2005). The Nash–Sutcliffe coefficient improved from an initial value of −25.7 for the cold simulation to +0.32 after calibration for the period 1997–2004 (Fig. 2b).

2.5. Modelling pesticides

Pesticide data were collected by CODA (2003) by taking daily grab samples of river water. Furthermore, inquiries were conducted during the spring seasons of 1998 until 2001. The farmers were asked to give as detailed information as possible concerning the amount of pesticide they applied, the application dates, the kinds of pesticides they used for their different crops and the treated surface. Forty-two percent of the farmers could give detailed information concerning the application dose and the day of application. In this study, we focus on the use of atrazine on corn (15% of the area) during the growth season of 1998, when the application rate amounted to 0.741 kg/ha. As only the treated fraction of corn fields is known for a certain day of application and not the exact fields, this fraction was taken from the total application rate and applied on all corn fields. As such, in reality we can expect higher concentrations at a certain time in a certain reach in the catchment than what is simulated in this homogenised approach. This approach is acceptable in the absence of detailed field data and in the lumped HRU approach of the SWAT model. The validation of the model for predicting pesticide fluxes is performed for the period 1999 to 2002.

3. Model improvements for pesticide fluxes

3.1. Original source code

By adding pesticide characteristics and management to the SWAT-model, the movement of the chemical in the watershed could be predicted. A comparison of simulated and measured pesticide concentrations in solution at the mouth of the river showed that runoff related pesticide peaks could be modelled but needed further calibration. Most importantly, the SWAT2000 model could not represent pesticide peaks during dry periods, a clear model deficiency. This is represented for the year 1998 in Fig. 3. The figure shows that pesticide peaks originating from direct losses are missed.

3.2. Implementing direct losses

To enable the SWAT code to account for direct losses, the codes were slightly modified by changing the parameter ‘AP_EF’ (application efficiency coefficient). Originally, the parameter indicates the process whereby a fraction of the applied rate is lost from the catchment. In the adapted code, the parameter is changed to the process by which a fraction of the applied pesticide is diverted directly to the river system, i.e. a direct loss. The adaptation of the source code existed in a modification in the module for pesticide application, which can be written as:

$$direct\_loss_{point} = ap_{rate} \cdot (1 - AP\_EF) \cdot area_{hru} \cdot 1e8 \qquad (5)$$

where $direct\_loss_{point}$ is the amount of pesticide lost during or immediately after application as a point loss (mg); ap_{rate} is the pesticide application rate (kg/ha); AP_EF is the pesticide application efficiency; $area_{hru}$ is the area of the HRU (km^2) and $1e8$ is a unit conversion factor. The effective amount of pesticide applied on the field then becomes:

$$pest2 = pest \cdot AP\_EF \qquad (6)$$

where $pest2$ is the effective amount of pesticide applied (kg/ha) and $pest$ is the actual amount of pesticide applied (kg/ha).

The direct losses are summed to the outputs of the land phase section in SWAT and are in that way directly diverted to the river. These modifications are allowed in this case, in which pesticides are not applied by airplanes but directly on the fields by spray equipment and where losses outside the system are not expected to be significant. The direct losses are considered to be lower than 2%, but their impact is

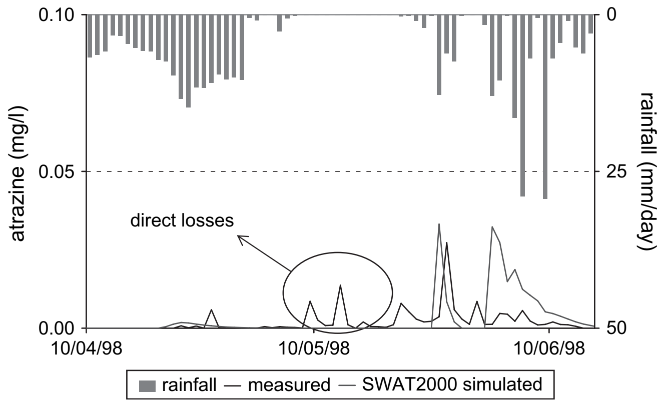

Fig. 3. Measured and predicted atrazine concentrations in the Nil (1998), with missing direct losses.

nevertheless significant since these losses are directly ending up in the river system.

The result of the implementation of direct losses can be seen in Fig. 4. Direct losses are simulated, but the simulation results clearly overpredict the pesticide concentrations with an estimated order of magnitude of 2.

3.3. Setting the mass balances right

After applying the above mentioned change, mass balances could be checked. If the application efficiency *AP_EF* was set to 0, all applied pesticides should reach the river by direct loss and only a little fraction is expected to be lost during transport in the reach. To assess this, a closer look was given to Subbasin 25 with atrazine application. It seemed that the applied dose reached the river, but during transport through the river almost half of the applied pesticide dose disappeared. The relevant mass balances are presented in Table 1. In a first attempt to elucidate the origin of the problem, the in-stream pesticide processes were deactivated, such as adsorption/desorption to sediments. This change did not affect the errors in the mass balance.

Apparently, SWAT only considers chemicals when flows equal at least 0.01 m^3/s. Since point source pollution typically happens during low flow periods, and because the model was subdivided into hundreds of sub-units (HRUs), this threshold was not always reached. Therefore, this threshold was reduced in several routines of the code to the value of 0.000001.

The mass balances represented in Table 1 show that the abovementioned modifications result in more realistic values for pesticides in solution, both at reach-level and at the mouth of the river. At the mouth of the river, there is still a small amount of pesticides missing.

Concerning pesticides sorbed on suspended solids, no sorbed pesticides could be found when the application efficiency *AP_EF* was set to 0, as in the source code direct losses were sent to the river as solubles and all river processes (including sorption) were de-activated. If the application efficiency *AP_EF* was given a value different from 0, lowering the

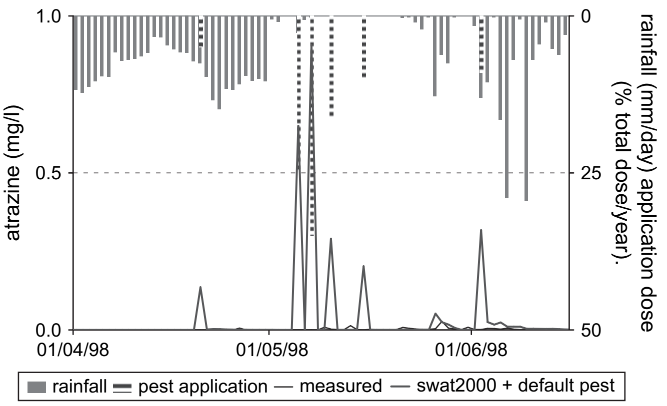

Fig. 4. Measured and predicted atrazine concentrations at the mouth of the river Nil (1998), with contribution of direct losses.

Table 1
Simulated amounts of pesticides in solution leaving different locations in the Nil catchment during the year 1998 with an application efficiency set to 0 (*AP_EF*), for different versions of the SWAT2000 source code

		Sub-basin 25	Reach 25	At the mouth
Area of corn (ha)		37.5		569.7
Applied dose (kg/ha)		0.741		0.741
Expected load (kg)	=area × dose	27.78	27.78	422
Without modific. (kg)	=initial model	0	0	0
Modification 1 (kg)	=implementation direct losses	27.75	**0.11**	**214.98**
Modification 2 (kg)	=correction of flow threshold	27.75	27.75	**419.29**
Modification 3 (kg)	=correct. mass losses/creation	27.75	27.75	421.47

Strange values are represented in bold.

threshold value of flow for chemical routing resulted in a simulation of sorbed pesticides too (results not shown).

3.4. Corrections for losses or mass creation

A closer look at the losses from and inputs towards the river revealed that there were some miscalculations in the original source code. Therefore, the following modifications were performed:

(1) The losses in the river were originally based on the basis of the residence time calculations, while the integration time step should be used.

In order to get stable calculations in the SWAT routing, both the Muskingum method and the Variable Storage method were not applied in a correct way. In both methods, the hydrological state variables such as river flow, depth, wetted perimeter, cross section area and flow velocities are based on the summation of the water volume in the reach and the inflow instead of taking only the volume:

$$A_{ch} = \frac{(V_{in} + V_{stor})}{L_{ch} \cdot 1000} \tag{7}$$

where A_{ch} is the cross-sectional area of flow in the channel (m^2) and L_{ch} is the length of the channel (km). This gave stable calculations also for small reaches because the inflow acts as a kind of a buffer for the storage in the reach, but the stability has wrong underlying calculations. In case the residence time is an hour, 24 times of the volume is added to the reach volume and the resulting calculated travelling time *TT* is a strong underestimation of an order of 24, while the river depth calculation is strongly overestimated. When the Muskingum and Variable Storage routing would be implemented in the proper way, unstable river volume and hence river depth and velocities would be obtained in situations with short residence times. Therefore, a new routing module was developed in which the relation between velocity and flow is calculated by solving of the Manning equations in an iterative way until an A_{ch} is found (and corresponding R_{ch}) that corresponds to the inflow q_{ch}:

$$q_{ch} = \frac{A_{ch} \cdot R_{ch}^{2/3} \cdot slp_{ch}^{2/3}}{n} \quad (8)$$

where q_{ch} is the flow rate in the channel (m^3/s), A_{ch} is the cross-sectional area of flow in the channel (m^2), R_{ch} is the hydraulic radius for a given depth of flow (m), slp_{ch} is the slope along the channel length (m/m) and n is Manning's 'n' coefficient for the channel.

A corresponding reach volume $V_{manning}$ can be calculated:

$$V_{manning} = 1000 \cdot L_{ch} \cdot A_{ch} \quad (9)$$

The key solution to get stable calculations for both situations where the residence time is smaller or bigger than the calculations is to use distinct equations for these situations. In case the residence time γ (days) is smaller than the calculation time step Δt (days), the reach volume at the end $V_{stored,2}$ (m^3) of the time step will be equal to $V_{manning}$ (m^3):

$$V_{stored,2} = V_{manning} \quad (10)$$

However, when γ is larger than the time step, only part of $V_{stored,1}$ (m^3) will be replaced by $V_{manning}$:

$$V_{stored,2} = V_{stored,1} - \frac{\Delta t}{\gamma}\left(V_{stored,1} - V_{manning}\right) \quad (11)$$

The routing component needed a final correction: the calculation of the transitional losses (infiltration or evaporation) and the river bank contributions are based on the calculation time step in stead of the residence time. Note that they are also heavily influenced by the previous routing corrections as they depend on the wetted perimeter too.

(2) For the routing of water the addition of bank flow is included. This water is added to the river outflow, but the concentrations of the chemicals were in the meantime kept constant whereas a dilution factor should be applied. This resulted in a creation of pesticide mass. This was corrected by calculating chemical concentrations based on the total amount of water leaving the reach.

(3) Finally, in the original source code the losses in the river bed were abstracted from the available water, but the concentrations were kept constant. This resulted in a loss of chemicals/pollutions. This approach was kept for the solutes, as they leave the system with the infiltrating or evaporating water. It was programmed that the solids remain in the river during the evaporation/infiltration, resulting in an increase in their concentrations.

The mass balance results of these modifications are presented in Table 1 and show that reliable mass balances are now achieved. If all the applied pesticides are assumed to be direct losses ($AP\_EF = 0$), the simulated load of atrazine at the mouth of the river in the year 1998 amounts 421.47 kg in case all processes and losses are ignored. This is in good agreement with the applied dose of 422 kg.

3.5. Implementing an estimator for drift

During the application of plant protection products, a part of the spray liquid may be carried out of the treated area by wind or the air stream of the sprayer and reach a nearby river system. Therefore, an estimator for drift was added to the source code, in order to estimate the contribution of drift to the direct losses.

The calculation of spray drift deposition was based on the German drift database (Ganzelmeier et al., 1995). These data were generated from a series of studies (at a number of locations and with a variety of crops) whose objective was to determine the absolute level of drift in practice under a variety of conditions. However, even this extended database partly reflects environmental, crop and application factors prevailing in Germany, but is recommended by the FOCUS Surface Water workgroup because it is currently the most comprehensive, widely available data set. The use of this database also has significant precedent in the EU evaluation process (FOCUS, 2001). The database is useful for plant protection products that are applied in compliance with the principles of Good Agricultural Practice. This comprises the application during low wind velocities only, the use of approved equipment as well as the application under favourable climatic conditions only. If these rules are not observed, it must be expected that larger amounts of plant protection products will be drifted than specified by the basic drift values. Ganzelmeier drift data are the 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 good agricultural practice. As such, they have to be considered as conservative.

The mean (integrated) drift deposition into surface water bodies can be calculated from the following equation:

$$\overline{Drift} = \left[A \cdot \int_{z_1}^{z_2} \left(z^B\right) \mathrm{d}z\right] \cdot \frac{1}{z_2 - z_1} \quad (12)$$

where $\overline{Drift}$ is the mean percent drift loading across a water body that extends from a distance of z_1 to z_2 from the edge of the treated field (%); A and B are previously defined regression parameters (Ganzelmeier et al., 1995); z_1 is the distance from the edge of the treated field to the closest edge of the water body (m); and z_2 is the distance from the edge of the treated field to the farthest edge of the water body (m). Here it is possible to take into account the effect of buffer zones on the reduction of drift towards a river. In this case, the values for z_1 and z_2 will be increased with the width w of the buffer strip.

The integrated form of this equation is as follows:

$$\overline{Drift} = \frac{A}{(z_2 - z_1) \cdot (B+1)} \cdot \left[z_2^{B+1} - z_1^{B+1}\right] \quad (13)$$

The values for A and B were extracted from the database of Ganzelmeier et al. (1995). As the focus is on the application of atrazine on corn fields, values for 'arable crops' with

'3 applications' were looked for. The values found for A and B are respectively 2.0244 and −0.9956.

To calculate the drift loading towards the receiving surface water, the following formula was implemented:

$$direct\_loss_{drift} = ap_{rate} \cdot AP\_EF \cdot area_{river\_reach} \cdot \overline{Drift} \quad (14)$$

where $direct\_loss_{drift}$ is the amount of pesticide lost during application as a drift loss (mg); ap_{rate} is the pesticide application rate (kg/ha); AP_EF is the pesticide application efficiency; and $area_{river\_reach}$ is the area of the receiving water (m^2); $\overline{Drift}$ is the mean percent drift loading across a water body that extends from a distance of z_1 to z_2 from the edge of the treated field (%). The area of the receiving water is calculated as follows:

$$area_{river\_reach} = \sqrt{area_{hru}} \cdot W \cdot 1000 \quad (15)$$

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 root square of the HRU area. As in SWAT the HRUs are lumped areas, no information about the position of the original fields is available. Therefore, as it is not known if a field borders the stream according to its length or to its width, the assumption that HRUs have a squared shape forms an intermediate solution.

The effective amount of pesticide applied on the field can then be calculated by subtracting point losses, drift losses to the river and the amount of drifted pesticide that is captured in the buffer zone, 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. If more detailed information concerning the application conditions is available, more detailed drift descriptions could be used for simulation (Cox et al., 2000; Gil and Sinfort, 2005; Tsai et al., 2005).

4. Results and discussion

After the model was improved for direct losses, a calibration of the most sensitive parameters (Holvoet et al., 2005) was performed for the year 1998, followed by a validation for the years 1999–2002. In Fig. 5, measurement and simulation results for atrazine concentrations are presented for the spring periods (1st of March until the end of July) during the years 1998–2002. As can be seen from Fig. 5, a good approximation between measured and simulated atrazine concentrations at the mouth of the river could be achieved. For the year 1998, the Nash–Sutcliffe improved from a value of −2.63 to 0.66.

If application data of pesticides are missing or incorrect (e.g. April–May 1999 and 2001), good simulations of direct losses are obviously impossible. Moreover, predicted values of direct losses will always be rough estimates, as currently the application efficiency AP_EF has a constant value during simulation. In reality, the efficiency will be highly variable over the different applications, due to variability in farmers, in farmers working methods and in daily differences. Nevertheless, this approximation is much more correct than ignoring direct losses and results in more realistic mass fluxes. For the period 1998–2002, a value of 99.8% for the application efficiency AP_EF was found to result in a good average agreement between measured and predicted direct losses.

During the years 2000–2001, sensitization of farmers resulted in a significant decrease of pesticide loads in the river. For atrazine, only during 2001 a reduction took place (Beernaerts et al., 2002). This reduction could also be seen during calibration: a higher value for the application efficiency AP_EF of 99.9% resulted in a better fit during 2001, whereas a lower value could better predict direct losses during the remaining years.

Special attention should be paid to the simulation of hydrology, as concentrations of direct losses are based on the mass of water passing the system. If flows are underestimated, pesticide concentrations will be over predicted and vice versa.

In Fig. 6, the contribution of drift to the total losses is represented for the load coming from sub-basin 25 towards reach 25. Sub-basin 25 consists of many corn fields on which atrazine is applied. In Fig. 6a no buffer zone was considered, whereas in Fig. 6b a 'no spray zone' of 1 m was introduced which matches the Good Agricultural Practise in Belgium. From these figures, it can be deduced that even a small 'no spray zone' has positive effect on pesticide mass fluxes towards a river system. The drift losses could be reduced by 87%. On application days, the fraction of drift towards the river was reduced from 13% of the application dose per unit area towards 1.6%, with respectively standard deviations of 0.057 and 0.007. The latter fraction is in agreement with values found in literature, stating that current drift losses towards an adjacent river in West Europe amount to between 1% and 2% of the dosage per unit area (de Snoo and de Wit, 1998; Siebers et al., 2003). Nevertheless, the contribution of drift losses to the total pesticide load in the river system is almost negligible. Bach et al. (2001) also found that the input in arable farming by spraydrift is very low. On the other hand, in fruit culture these contributions may be significant.

On the other hand, point losses are very important. On a yearly basis, point losses contributed for 30% up to 90% of the pesticide loss during the period 1998–2002 and therefore warrant special attention in pesticide reduction strategies. This is especially true because point losses occur during low flow conditions, which can result in severe impacts on water ecosystems. Most severe impacts can be expected in upstream, small rivers, where different amphibians, fishes, etc. brood and have their habitat.

As mentioned before by different authors (Dabrowski and Schulz, 2003; Leu et al., 2004b), the importance of runoff as a transport route of pesticides towards the river was also demonstrated in this modelling study. Different management strategies can reduce these fluxes (Mostaghimi et al., 2001; Schreiber et al., 2001; Santhi et al., 2006).

5. Conclusions

In this study, an attempt was made to include the description of direct pesticide losses in the source code of the

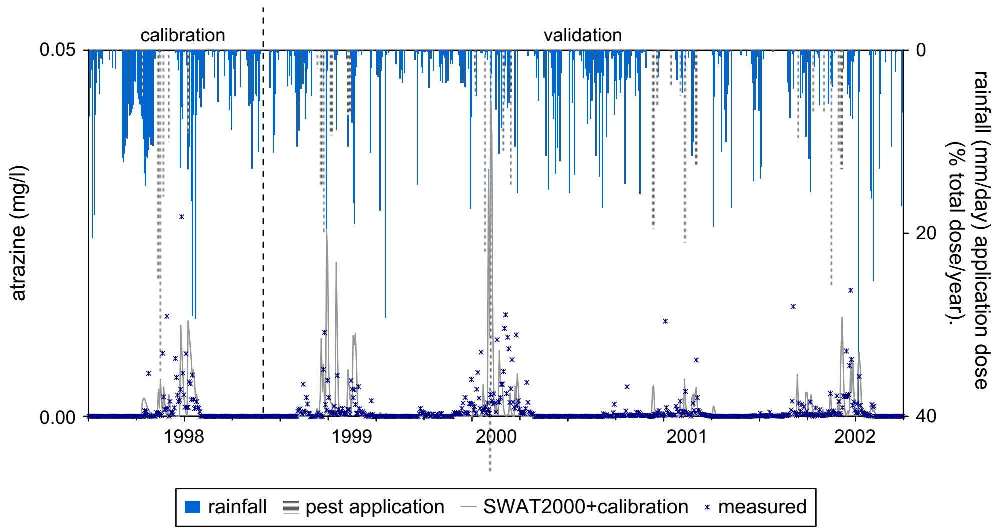

Fig. 5. Measured and predicted atrazine concentrations at the mouth of the river Nil after calibration (spring periods of 1998–2002).

SWAT model. Different steps were taken to improve model predictions for direct losses. If application data for pesticides are available and reliable, reasonable predictions can be made. Nevertheless, as the occurrence of point losses is subject to an enormous variability, only average estimates can be expected. It would be useful to describe application data as probability distributions, in order to reflect the uncertainty related to these parameters.

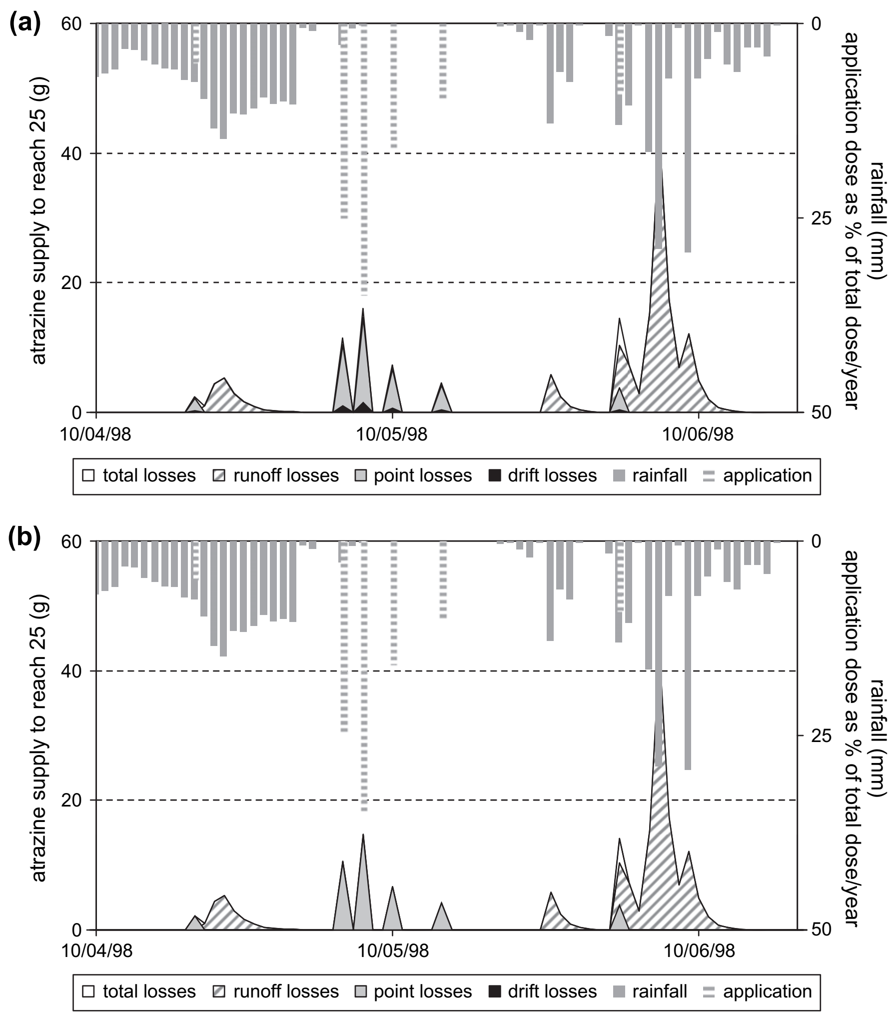

Fig. 6. Predicted atrazine loads coming from sub-basin 25 (a) in the absence of a buffer strip and (b) with a buffer strip of 1 m between field border and river.

It was shown that the lower atrazine load leaving the river Nil during the year 2001, could to a large extent be attributed to a higher engagement of the farmers: a higher application efficiency *AP_EF* during 2001 could describe the decrease of direct losses. Sensitization of farmers seems to play an import role in reducing direct losses. During the year 2001, the percentage of applied pesticide leaving with the river decreased from 2% in 1998 to 0.3% in the year 2001.

By means of a drift estimator, the contribution of drift in direct losses could be estimated. It was found that the contribution of drift losses to the total pesticide losses is of minor importance compared to point losses and losses coming from runoff. The latter need special attention in pesticide management strategies.

Former pesticide studies performed with the SWAT2000 model (Neitsch et al., 2002b; Winchell et al., 2005; Santhi et al., 2006) should be examined with caution, as errors were detected in the source code by checking mass balances. The errors were resolved.

The modified SWAT code can be used for quantification of pesticide reductions through different measures. Hereby, ranking of different measures based on effectiveness will be possible. Moreover, realistic risk assessments can be performed, taking into account spatial and temporal variability of pesticide applications on catchment scale.

Acknowledgements

We would like to thank M. Vanclooster and A. Romanowicz of the Catholic University of Louvain-la-Neuve for providing data on the Nil-catchment and for preparing the detailed land use map. We are grateful to CODA for making their monitoring data of the Nil-catchment and the results of their questionnaires available. The work of A.v.G. has been partially supported by the EU Harmoni-CA project, contract no EVK1-CT-2002–2003. P.V. is Canada Research Chair of Water Quality Modelling.

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