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# Engaging Stakeholders in assessing the impact of agricultural practice on groundwater quality: the Residence Time Distribution model (RTD)

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**Abstract:**
Nowadays, the quantitative evaluation of the impact of agricultural practices on groundwater quality and the estimation of the underground residence time of contaminants can be achieved by solving the advection, dispersion, and reaction differential equations. However, the application of these models at the aquifer scale is always difficult because it requires a large amount of data to be consistent with field observations. Thus, in practice numerical modelling is mostly incompatible with the financial support of water managers and the time-frame imposed by the authorities for groundwater vulnerability assessments.
This note presents an operational approach to model contaminant migration from soil to drinking water boreholes. The proposed model used the residence time distribution concept and was designed to help stakeholders assess the quantitative impact of agricultural policies on underground water quality. First, the model is built by an expert, to define the underground properties describing underground pollutant migration. The model can be calibrated with historical datasets. Then an interface is proposed to enable stakeholders to estimate the impact of a future agricultural practice on the water quality. These estimations are based on scenarios of contaminant release (such as nitrate flux under the crops) defined and applied on the different vulnerability areas delimited on the watershed by the stakeholder himself.
The interface allows stakeholders to interactively define the areas of vulnerability. The possibilities of this approach are presented here through the example of the "La Saussaye" borehole water supply draining the Beauce limestone aquifer (Chartres, France), where nitrate has increased since the 1980s.

***Keywords**:*
Diffuse pollution; groundwater contamination; nitrate; drinking water borehole; management tools.

## 1. INTRODUCTION

The main tools for the management and the conservation of groundwater resources consist in characterizing the vulnerability of the aquifer used for drinking water. Intrinsic vulnerability uses physical characteristics as criteria to determine the sensitivity of groundwater to surface pollution. Most intrinsic vulnerability maps are multi-criteria, weighted and index-based, developed by Aller et al., (1987), Doerfliger et al., (1999). These tools enable policies for the development of codes of practice for groundwater protection to be proposed (Escolero et al. 2002) and open up significant opportunities to enhance the efficacy of water vulnerability assessment tools by incorporating indicators and operational measures for social considerations (Plummer et al., 2012). While the area of use is huge, the index calculation method is limited because the weighting is usually arbitrarily chosen. These approaches are qualitative and highly subject to the hydrogeologist's interpretation (Panagopoulos et al., 2006).
To overcome these limitations, borehole vulnerability analysis was developed for watersheds supplying drinking water. This method completes the vulnerability index with the notions of distance, horizontal flow rate and transport to the target (borehole or spring) (Goldscheider and Popescu, 2003). Two types of vulnerability can be defined: a resource vulnerability which only takes vertical transfer into account, and a borehole vulnerability which incorporates horizontal transfer into the borehole.
For all these methods, developed to validate vulnerability criteria, the aim was to link surface land use with watershed hydrodynamic properties and water quality at the boreholes. Using time series analysis, several studies have applied an impulse response at the watershed scale for solute transport modeling purposes (Jury, 1982; Beltman et al., 1994; Molénat et al., 1999). The method consists in establishing a residence time distribution (RTD) to link contaminants at the surface of the watershed to the contaminant concentrations measured in the borehole.
Linking the spatial properties that determine the vulnerability and the temporal evolution of the water quality is a key point for water resource management. At the watershed scale, some semi-distributed models incorporate the soil surface properties to model water quality with a GIS dataset based on an impulse response, such as the SWAT model (Srinivasan and Arnold, 1994) or on a flow model such as Drainmod (Fernandez et al. 2006) MACRO (Larsbo and Jarvis, 2003) and STICS (Ledoux, 2003). For groundwater quality purposes, the flow paths must be analyzed in 3 dimensions but few tools are available to compute an impulse response from the spatially distributed 3D groundwater properties.
This paper used the method proposed by Dedewanou et al. 2015, using the RTD impulse response in aquifers based on spatial datasets used for specific vulnerability assessments. Based on the impulse response, which is characteristic of the watershed, a vulnerability index is calculated for each agricultural crop and the groundwater quality can be estimated. After validation of the RTD with historical datasets of water quality at the borehole, the proposed approach enables stakeholders to use a friendly graphical user interface (GUI Windows®) to define the vulnerability areas and assess the impact of an agricultural policy on the water quality at the drinking borehole.
This GUI involves directly the stakeholders in the definition of vulnerability areas, which is new for groundwater management. Therefore, with this novel tool, the stakeholders can interactively test the impact of land use on groundwater quality (in a fixed hydrogeological context) allowing the management of conflicting stakeholder demands.

## 2. MODEL DESCRIPTION

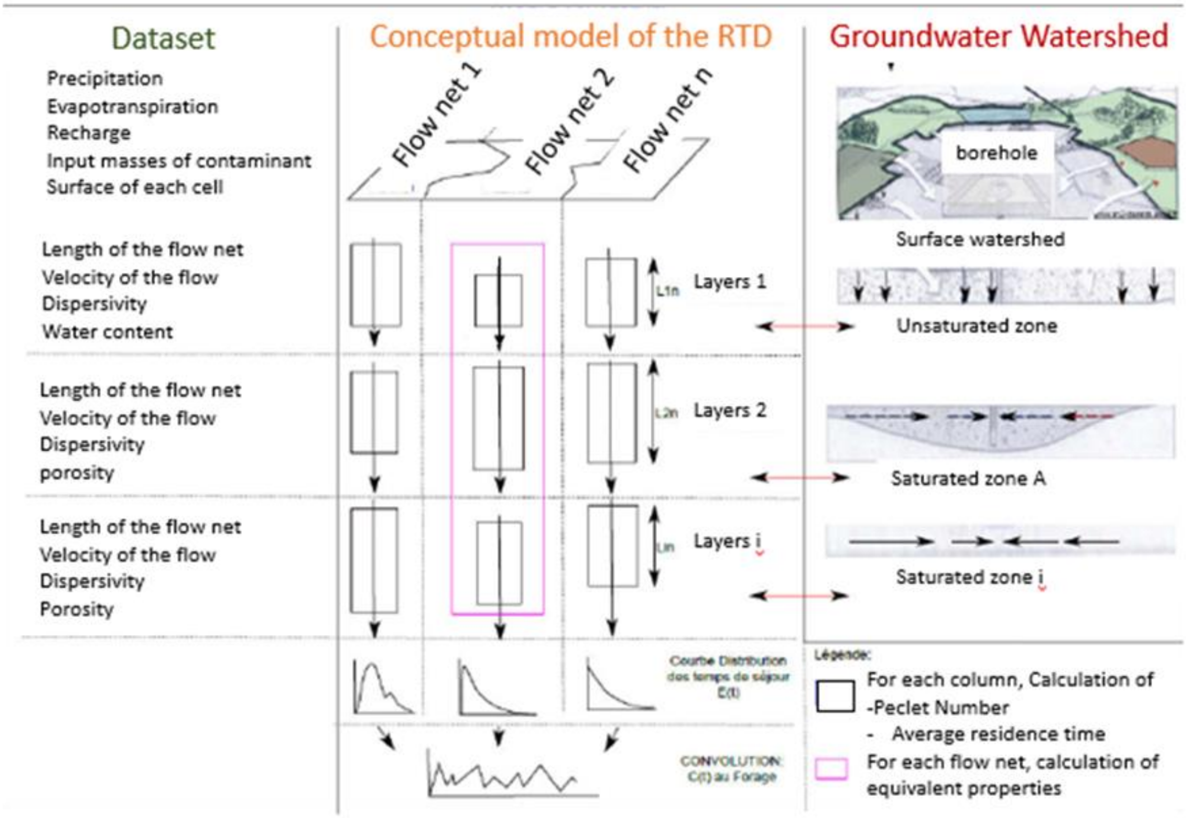


**Figure 1: Description of the model approach, with A/ the input dataset, B/ the conceptual model of i layers and n flow paths and C/ a scheme of the corresponding borehole catchment.**

The Residence Time Distribution (RTD) is a probability distribution function that describes the amount of time a fluid element can spend inside the rock for a pulse of mass M=1. In our approach the RTD is defined as:

$$E(t) = \sqrt{\frac{Pe}{4\pi t\bar{t}}}\exp\left[-\frac{Pe\,(\bar{t}-t)^2}{4t\bar{t}}\right]\exp\left[-\lambda\, t\right] \quad \textit{(Equation 1)}$$

The RDT use the non-dimensional Peclet number, the average residence time($\bar{t}$), and the rate of degradation $\lambda$ of the contaminant during transport. The contaminant transport through the rock column is described with only three parameters and no assumption is made on the laminar, turbulent, unsaturated or saturated nature of the flows. The concentration of contaminant at the borehole is calculated by a convolution between E(t) and the history of the incoming masses on the agricultural crop.

Underground transport is addressed by decomposing the watershed of the borehole water supply into *n* parallel flow nets (Figure 1). This reduces the non-linear three-dimensional problem to a linear one-dimensional one. The water and the contaminant mass which infiltrate the ground enter through the various $i$ unsaturated or saturated layers of the aquifer where the hydro-dispersive properties can vary. They flow through the $i$ layers until the borehole, following the flow nets. For every $n$ flow net, the $i$ layers are considered to be independent columns characterized by the average residence time $\bar{t}_{n,i}$ and the Peclet number $Pe_{n,i}$. For $i$ serial layers, equivalent properties can be computed.

The equivalent RTD for each flow net represents the mass arriving at the borehole for an injected mass equal to 1. Depending on the value of the equivalent parameters, and on the discharge Qn for the n th flow net, the concentrations obtained make it possible to identify flow nets showing concentrations higher than a threshold while other flow nets present concentrations below the threshold. The spatialized grid of equivalent parameters locates the surfaces which contribute to the over-concentration measured at the groundwater borehole, making it possible to prioritize the various surfaces in terms of borehole vulnerability and/or risk.

## 3. THE CHARTRES DRINKING WATER SUPPLY

The Chartres water supply is a borehole pumping 450 $m^3$/h of water for 39 000 inhabitants in the Séno-Turonien chalk aquifer of the Paris basin. The 60 km² catchment of this borehole is agricultural land where farmers use nitrogen addition to improve the production of wheat, rape and maize. The borehole water quality has shown a continuous increase in nitrates from the 1980s, now reaching 60 mg/L. Water managers have to mix this water with less polluted water in order to meet the 50 mg/L drinking water quality standard. In the present study, the aim of the RTD model is to provide stakeholders with a quantitative tool to optimize future agricultural practices with respect to the nitrate load of drinking water. The borehole water catchment was defined using the hydro-geo-chemical method described in Binet et al. 2006. Input data for the calculation of the RTD come from spatial databases, such as pedology, geology (www.infoterre.fr) or precipitation and evapotranspiration (www.meteofrance.fr) and contaminants below the root zone (Ledoux, 2003) and require GIS to estimate the equivalent lumped parameters.
The validation step was conducted using historical monitoring of the water quality below the root zone with nitrate flux from the 1980s to today, or with residence time tracers such as chlorofluorocarbon (CFC) change in the atmosphere (Long and Putnam 2006). These incoming time series were used as the upper boundary condition and the model estimated the water quality downstream, at the borehole, following the RTD of the contaminants. The confrontation between observed and modelled concentrations at the borehole was considered to be an indicator of the efficiency of the model.
The next step presented here is the development of a Windows interface for stakeholders that enables them to easily test the impact of agricultural policies on the watershed, such as for example reducing the incoming nitrate flux by about 10% in the highly vulnerable zone, on the water quality at the borehole.

## 4. RESULTS

### 4.1 Modelling the water quality at the drinking borehole

The RTD computed from the GIS dataset reduces the groundwater system to an input / output system. Using the data of surface nitrate quality or of atmospheric CFC the RTD model can estimate the change in water quality at the borehole driven by these changes in surface water quality. Figure 2 presents two examples of the model validation. The RTD estimated for the Chartres drinking borehole appears relevant to describe both residence time estimated with concentration of CFCs and SF6 observed at the borehole in November 2013 and nitrate evolution through time.

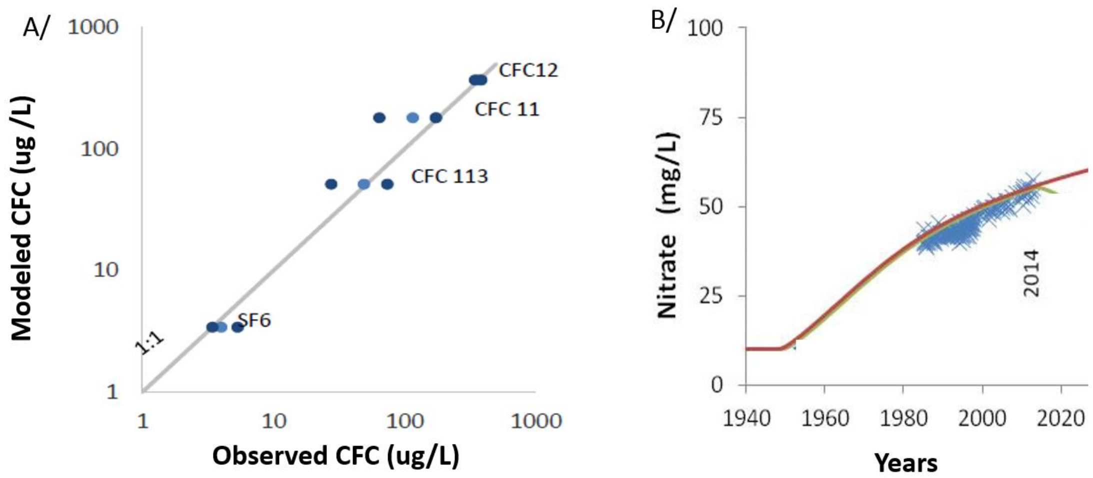


**Figure 2. Model validation with A/ Calculated versus modelled concentrations for CFC and SF6 sampled at the drinking boreholes in 2013 and B/ observed and modelled nitrate time evolution**

### 4.2. An interactive vulnerability map

The average residence time of water calculated with the RTD model of each cell of the watershed is represented on the x axis of the scatterplot in Figure 3. Average residence times range between 5 to 90 years. The y axis represents the maximum percentage of the mass that can reach the borehole in a time step (here one year). A high value means that the contaminant was not submitted to dilution and dispersion processes. A low value means that the contaminant from the cell i is diluted or dispersed before reaching the borehole. The vulnerability mapping consists in dividing the watershed into cells and assigning a vulnerability class to each cell, represented by the green-blue-orange-red color scale on the map in Figure 3. Each vulnerability area can be defined by the user moving the four colored squares on the RTD curve. High vulnerability cells correspond to cells with a short residence time and low dispersion/ dilution effect.

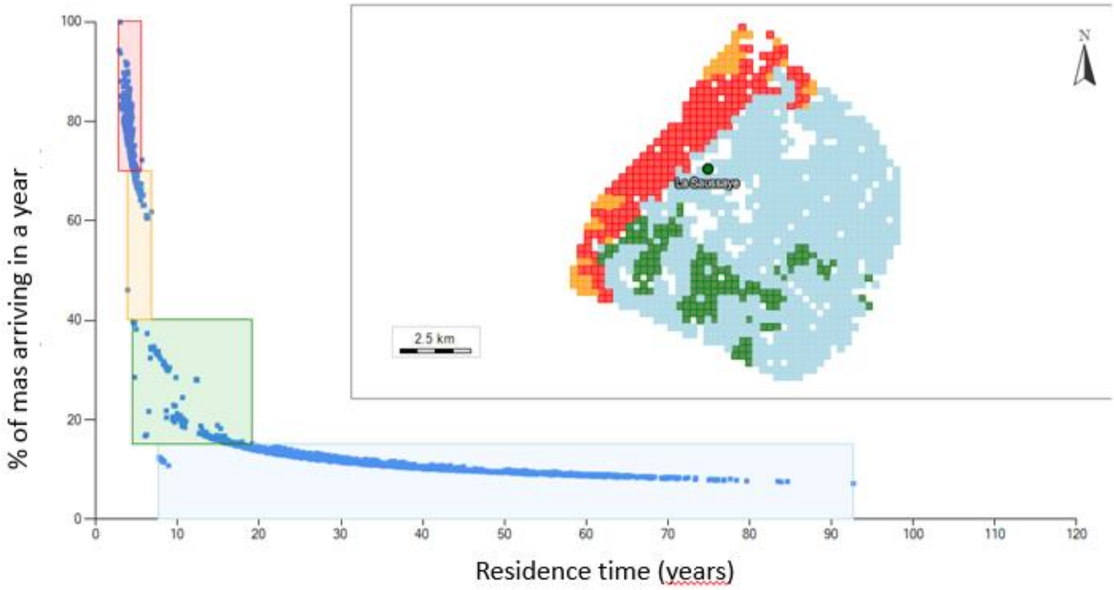


**Figure 3: Visualization of the interface for the vulnerability mapping. The blue-green-orange-red colors shows the vulnerability index, defined with the residence time and the % of mass arriving in a year. The vulnerability of each parcel of the borehole watershed is presented on the map.**

### 4.3. An interface to estimate the impact and delay of agricultural policy on the water quality at the borehole

The current scenario is defined as the scenario that explains the evolution of the nitrate increase from the 1980s up to the present. This is the reference scenario used for model calibration. The interface presented in Figure 4 enables the impact of a decrease in the inflowing nitrate on the water concentration observed at the borehole to be estimated. Each scenario is represented by a percentage of change compared to current practice and each scenario has a different impact on the water quality. For example, if water managers decide to reduce the nitrate input on the low vulnerable area by about 10%, it will take 20 years to stabilize the nitrate increase, because of the high residence time (yellow curve in Figure 4). In contrast, a 30% decrease in the nitrate input in the highly vulnerable area will create a significant decrease in nitrate at the borehole, but the increase will pick up again because the observed increases are driven by the low vulnerable zone (green line).

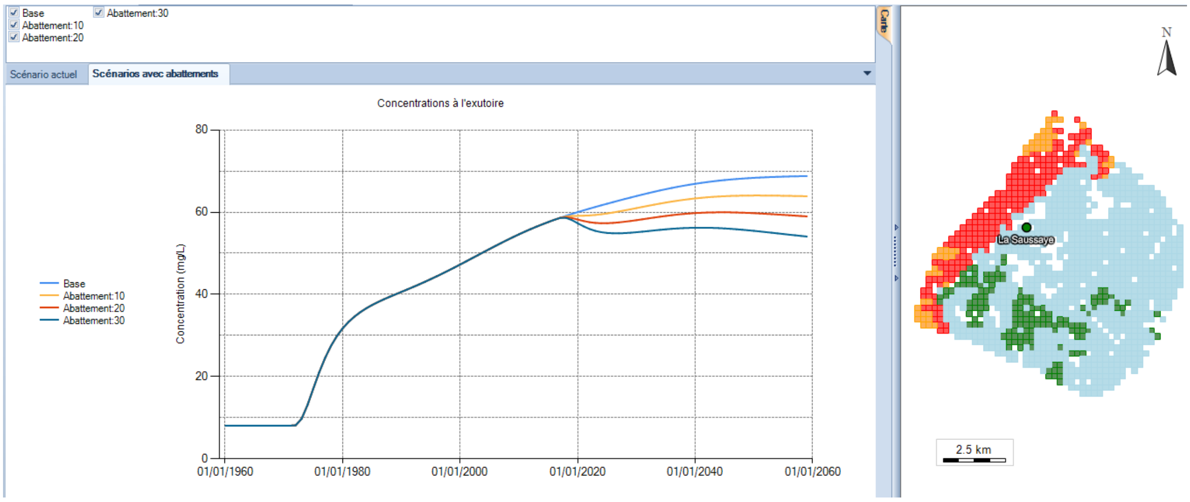


**Figure 4: Visualization of interface to assess the impact of agricultural policy on nitrate evolution at the borehole. 4 scenarios for the same practice (in blue) are presented, reducing the nitrate load on the overall watershed by about 10%, 20% and 30%. The scenarios can be different for each vulnerability zone.**

Thus, the interface is designed to enable easy visualization of the spatio-temporal change in the water quality and can be easily used by all the stakeholders. No intervention by the scientist is required to run new scenarios and the model can become a tool for discussion between farmers and water managers on how to optimize nitrate use w.r.t. agricultural needs and water protection.

## 5. DISCUSSION AND CONCLUSION

The Residence Time Distribution model can address temporal and transient aspects of contaminant spreading and represent them in a semi-quantitative manner. Such an approach makes it possible to establish a spatial risk or vulnerability index validated by water quality changes at the borehole. Once the model has been developed, the risk mapped allows stakeholders to test the efficiency of land practice scenarios on the quality of the groundwater catchments without the intervention of an expert in groundwater contamination. In this approach, the hydrology was assumed to be steady state. Although many authors point out that water exchanges between the flow nets, this strong hypothesis was made for large time steps, such as years. In these conditions, it is preferable to describe the average behaviour of the system, which is easier to use for risk assessment. High or low water stages can be estimated from the extreme discharge values. The impact of global change, with changing recharge rate is not taken into account and this will be the next step to be implemented in our approach.
The model provides an estimation of the global amount of nitrate that agriculture needs to reduce in order to stabilize the contaminant increase at the borehole. The mapping approach enables land management to optimize the location of more sensitive surfaces in terms of groundwater quality. Lastly, the interface is a pedagogic tool for the co-construction of practices that are acceptable in terms of agricultural efficiency and in terms of groundwater management.
The interface allows stakeholders to run new scenarios to evaluate the impact of the agricultural practices on the water quality without involving directly the scientists. Therefore the tool could be easily re-used after the end of the scientific project, for instance in order to evaluate the impact of change of management policy, land practice, land-owner on the water quality.

## 6. ACKNOWLEDGMENTS

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