

Groundwater vulnerability assessment using physically based modelling: from challenges to pragmatic solutions

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Abstract Numerous groundwater vulnerability and risk mapping techniques have been developed taking into consideration a variable number of factors. The most common techniques are based on calculation of an index expressing the protective effect of underground formations overlying the groundwater resource. The limitation of most of these methods is related to their use of a qualitative definition of groundwater vulnerability, as opposed to a definition based on a quantitative description of contaminant migration. A physically-based point of view and definition of the vulnerability is proposed and based on three factors describing a pollution event, which are the contaminant transfer time from the hazard location to the 'target', the contamination duration at the 'target' and the level of contaminant concentration reached at the 'target'. This concept allows a clear distinction between conventional aspects and physically-based results in the building of a final vulnerability indicator. This methodology has the further advantage to consider the possible impact of runoff conditions occurring at the land surface and possibly leading to lateral contamination of groundwater through downstream preferential infiltration features. Practically, this method needs to describe and simulate the pollutant migration in the unsaturated zone and possibly in the saturated zone in order to assess the breakthrough curve at the 'target'. Preliminary application is illustrated on a case-study located in a limestone basin in Belgium. Perspectives are proposed towards a generalisation of the vulnerability concept for risk assessment within a pressure - state - impact framework.

Keywords groundwater vulnerability; susceptibility; sensitivity; contaminant migration; modelling; preferential infiltration; risk assessment; groundwater protection.

INTRODUCTION

Vulnerability, broadly defined here as the degree to which human and environmental systems are likely to experience harm due to a perturbation or stress, can be conceptualised in very different ways by scholars from different knowledge domains, and even within the same domain (Füssel, 2007). The term 'vulnerability' is used in quite different policy contexts, referring generally to different systems exposed to different kind of hazards. Unfortunately, a common term may still hide divergent assumptions. One can consider vulnerability of a specified system to a specified hazard

or range of hazards (Brooks, 2003). But some are arguing that vulnerability should focus instead on assessing the vulnerability of selected variables of concern and to specific sets of stresses (Luers *et al.*, 2003). In a risk – hazard framework, the vulnerability definition (terms as ‘sensitivity’ and ‘susceptibility’ were also used in the past) refers to a physical system. The state of this system is quantitatively estimated by variables of concern. A good starting point will consist here in specifying the chosen applied vulnerability concepts dealing with groundwater issues. In the groundwater community, the concept of vulnerability has considerably evolved from the first definitions (Albinet & Margat, 1970; US EPA, 1993; Vrba & Zaporozec, 1994). The EU COST Action 620 (2003) proposed an adequate and very logic terminology based on the work of Brouyère *et al.* (2001) and described recently by Frind *et al.* (2006). An intrinsic vulnerability is defined as the vulnerability of the groundwater to contaminants, taking into account the inherent geological, hydrological, and hydrogeological characteristics, but independent of the nature of the contaminants, while specific vulnerability additionally takes into account the chemical behaviour of the contaminant and the vulnerability of the groundwater to a particular contaminant or group of contaminants. An entire risk assessment procedure (Fig. 1) can then be considered (Brouyère *et al.*, 2001) as a hierarchical process starting with intrinsic vulnerability, then progressing to specific vulnerability, and finally to risk assessment when combining with hazard (i.e. potential pollution at the surface).

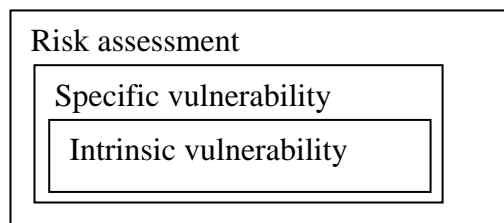


Fig. 1 Successive steps of a risk assessment procedure to groundwater contamination.

This approach is also based on the hazard-pathway-target model, which distinguishes between the groundwater ‘resource vulnerability’ and the ‘source vulnerability’ (Brouyère *et al.*, 2001). For ‘resource vulnerability’, the target is the saturated zone of the aquifer or the water table, and only the vertical path through the overlying layers is considered, while for the mapping of ‘source vulnerability’, the target is the well or the spring, and both the vertical path to the aquifer and the horizontal pathway within the aquifer must be considered. In this last case, the final aim is clearly to protect the well/spring, so that the complementary or overlapping role of ‘source vulnerability’ maps with the ‘well head protection area’ (usually based on the time a contaminant will take to reach the well) must be studied.

Until now, the most common approaches are overlay and index methods (Gogu & Dassargues, 2000) whereby the protective effect of the overlying layers is expressed in a semi-quantitative way. The various physical attributes of the system (i.e., geology, soil texture, depth to water table) are overlaid, with the help of a GIS. Weighting and rating are arbitrarily given and a final classification of the obtained vulnerability index makes possible to produce very nice coloured maps. Among many others, methods such as DRASTIC (Aller *et al.*, 1987), SINTACS (Civita & De Maio, 1997), EPIK (Doerfliger et al., 1999), PI (Goldscheider et al., 2000) are belonging to this category.

Each year, new similar methods are proposed in the literature because they are relatively easy to implement and require little data. However, the results can be questioned because these methods rely more on the judgment of the analyst than on the actual hydrogeological processes (Brouyère *et al.*, 2001; Frind *et al.*, 2006). Moreover, it has been shown that different overlay and index methods applied to the same system can yield dramatically dissimilar results (Gogu *et al.*, 2003).

PHYSICALLY-BASED GROUNDWATER VULNERABILITY

When trying to quantify the potential risk of pollution for a considered target, we support the idea that the following practical questions (Fig. 2) should be addressed: ‘if a pollution is likely to occur somewhere in the catchment, how long does it take to reach the target, and if so, to which extent and for how long could the target be polluted?’ (Brouyère *et al.*, 2001).

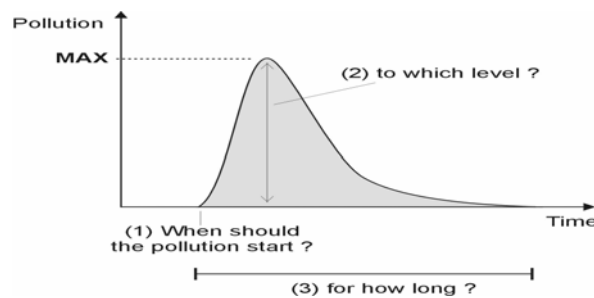


Fig. 2 Three proposed criteria for assessing vulnerability (Brouyère *et al.*, 2001).

The vulnerability assessment can then be based on the impulse response at the ‘target’ to a Dirac-type solicitation (point, unit mass, instantaneous source of pollution), considering only physical hydrodispersive processes for intrinsic vulnerability and both physical and biochemical processes for specific vulnerability. The breakthrough curve obtained after a vertical transfer through the overlying layers (Fig. 3) can be computed pixel by pixel. However, in the reality, there is a direct impact of a contamination on a target (i.e. contamination by direct vertical infiltration) and a lateral impact (i.e. contamination that infiltrates after a surface transfer) this last depending also on the surface hydrology conditions, topography, etc. This implies that, not only the mass of contaminant, but also the way it is distributed on land surface must be considered.

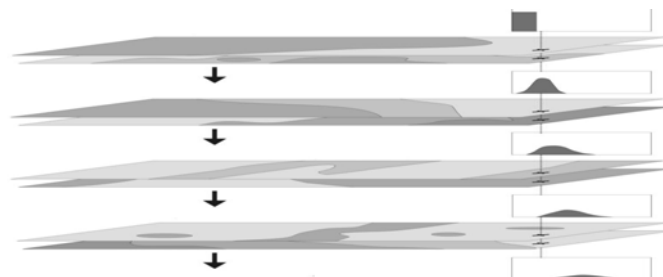


Fig. 3 Breakthrough curve changes during the pollutant transfer from the surface to the target (i.e. the water table).

This surface distribution of the infiltration can be computed by different ways and existing softwares. On the basis of runoff coefficients based on land use, slopes, and soil properties, Popescu *et al.* (2004) proposed an original method for quantifying a lateral ‘dangerosity’ coefficient taking into account how easy infiltration can occur in the considered pixel for a contaminant originating from another surface location in the catchment. This lateral ‘dangerosity’ accounting for the lateral impact of a contamination is combined to a direct ‘dangerosity’ accounting for the direct impact. The normalised total ‘dangerosity’ multiplies then, for each pixel, the results in terms of transfer time, duration and maximum concentration as obtained from 1D unsaturated-saturated flow and solute transport multi-layer simulation (for a Dirac input). At the end of the process, three maps are available: one for each of the physically based criteria defined here above (Fig. 2). Recent developments in modelling the unsaturated zone take into account the possible influence of epikarst, macropores (among others: Beven & Germann, 1981; Chen & Wagenet, 1992; Therrien & Sudicky, 1996; Brouyère *et al.*, 2004). In practice, large uncertainties can remain and are linked to the used values of the flow and solute transport parameters. However, this proposed methodology leads to maps with physically consistent information. It is only in a next step that decision makers can decide about the relative importance for each of the three criteria (i.e. transfer time, duration of the contamination, concentration) according to their locally agreed priorities, and then combining them in a weighted averaged index of vulnerability. This one will be used for defining classes of vulnerability according to the final aim of the study.

EXAMPLE OF APPLICATION

The Néblon basin (65 km²) is located in Belgium in the region of Condroz. Geologically, it belongs to the part of Devonian Carboniferous pleats formations of the eastern edge of the Dinant synclitorium that crosses Belgium from West to East. This region is characterised by typical alternation of shales and sandstones anticline crests and calcareous syncline depressions (Fig. 4). The geological formations are made of terrigenous detritical facies of Famennian age, carbonated rocks of carboniferous and terrigenous detritical sediments of Namurian age. Locally, ancient paleokarsts are filled by Tertiary sandy clay sediments. The region is also covered with loess formation. More descriptive information about hydrogeological characteristics of this zone can be found in Gogu *et al.* (2003).

Pixels of 30 m x 30 m have been used. On the basis of an extensive collect of data (Popescu *et al.*, 2004) a description of each 1D column from the soil surface to the water table has been obtained for each pixel. An algorithm has been developed for recognising columns with identical characteristics, so that the flow and solute computations for a Dirac input of contaminant are done only once for one type of column. Different maps are obtained for (a) transfer time, (b) duration of the contamination (above a given threshold) and (c) a normalised maximum concentration. On the basis of these results, any kind of vulnerability map can be built according to the weighting coefficients agreed by the local community or decision makers.

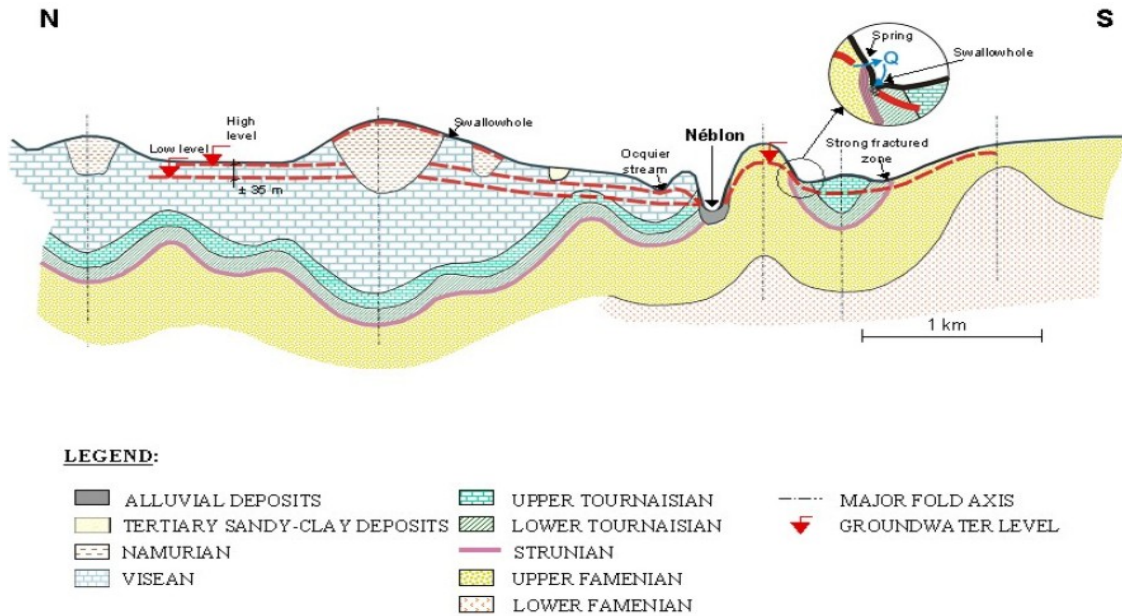


Fig. 4 Schematic N-S hydrogeological cross-section in the Néblon basin.

As an example, the vulnerability map of figure 5 shows the results obtained for a 0.45 weight given to the transfer time, 0.45 to duration of the contamination, 0.10 to the maximum concentration. A lot of combinations can be considered according to different purposes. It is important to point out that this method allows a clear distinction between the part of the analysis that is based on computing physically consistent processes and the purely empirical part (i.e. choice of the weighting coefficients and classes for obtaining different colours on the vulnerability map).

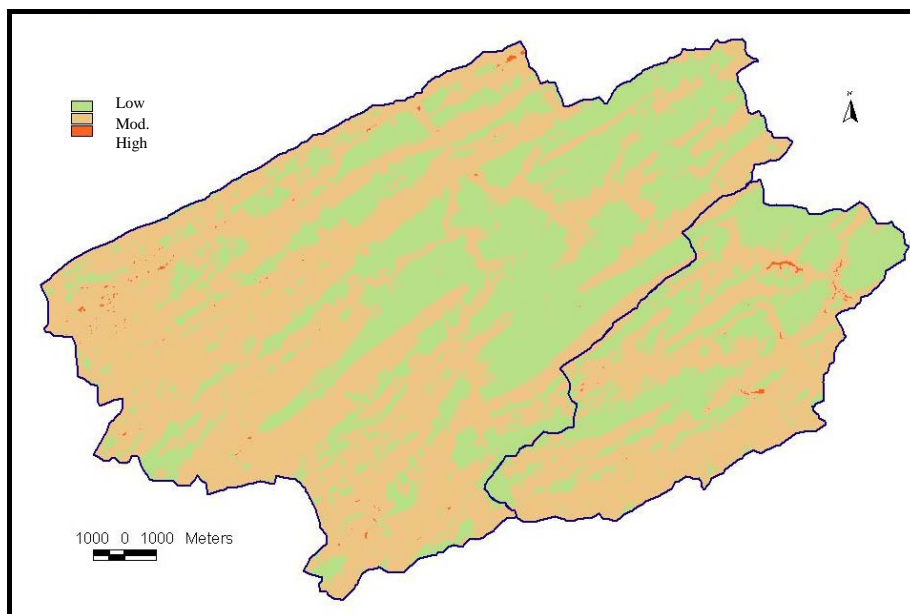


Fig. 5 Example of a vulnerability map for the Néblon basin as obtained for a 0.45 weight given to the transfer time, 0.45 to the contamination duration, 0.10 to the maximum concentration.

GENERALISATION AND PERSPECTIVES

Turner *et al.* (2003) give a definition for the vulnerability as the degree to which a system, subsystem, or system component is likely to experience harm due to exposure to a hazard. The method described here above is only a particular example where intrinsic vulnerability of the groundwater resource (as target) is considered. One can think to generalise the concept of groundwater vulnerability by including any kind of stress factor which can affect groundwater resources or any considered groundwater source. In this way, the proposed method is compliant with a Pressure-State-Impact (PSI) causal chain (Gardin *et al.*, 2006). For generalisation to any kind of stress factors, physically based criteria and indicators of changes can be proposed for the various subcomponents of the groundwater state as affected by pressures. Classical approaches such as DRASTIC are not suited to a PSI framework as they just produce colour maps not quantifying the importance of an impact given the importance of a pressure. In fact, the generalised concept of groundwater vulnerability should reflect the easiness with which the groundwater system (the 'state') transmits pressures into impacts. It consists in evaluating how a change in a given upstream factor (e.g. changes in groundwater recharge, surface contamination, etc.) has knock-on effects on downstream factors (e.g. base-flow to rivers, groundwater quality in a well, etc.). This turns to evaluating the following kind of derivatives (Gardin *et al.*, 2006):

$$V_{ij} = \frac{\partial DF_i}{\partial UF_j} \quad (1)$$

where V_{ij} is the vulnerability of the i^{th} downstream factor DF_i to a change in the j^{th} upstream factor UF_j . The larger V_{ij} the more sensitive is the groundwater state, in the sense that it will transmit more easily a pressure influencing the j^{th} upstream factor to an impact resulting from a change in the i^{th} downstream factor. Luers *et al.* (2003) proposed a very similar methodology applied to environmental problems in general. Of course the key steps consist in identifying the most appropriate modelling tools and methodologies for simulating the existing physical processes linking the various components of the PSI causal chain. It is here that the groundwater modellers community must be pro-active for providing the most adequate tools ranging from simple analytical solutions to spatially distributed physical models (surface and groundwater).

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