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A multi-scale method to assess pesticide contamination risks in agricultural watersheds

Francis Macary [a,*], Soizic Morin [b], Jean-Luc Probst [c,d], Frédéric Saudubray [a]

[a] *Irstea, UR ADBX Aménités et dynamiques des espaces ruraux, 50 avenue de Verdun, F-33612 Cestas Cedex, France*
[b] *Irstea, UR REBX Réseaux, épuration et qualité des eaux, 50 avenue de Verdun, F-33612 Cestas Cedex, France*
[c] *Université de Toulouse, INPT, UPS, Laboratoire Ecologie Fonctionnelle et Environnement (EcoLab), ENSAT, Avenue de l'Agrobiopôle, 31326 Castanet-Tolosan, France*
[d] *CNRS, EcoLab, 31326 Castanet-Tolosan, France*

ABSTRACT

The protection of water is now a major priority for environmental managers, especially around drinking pumping stations. In view of the new challenges facing water agencies, we developed a method designed to support their public policy decision-making, at a variety of different spatial scales. In this paper, we present this new spatial method, using remote sensing and a GIS, designed to determine the contamination risk due to agricultural inputs, such as pesticides. The originality of this method lies in the use of a very detailed spatial object, the RSO (Reference Spatial Object), which can be aggregated to many working and managing scales. This has been achieved thanks to the pixel size of the remote sensing, with a grid resolution of $30\,\text{m} \times 30\,\text{m}$ in our application.

The method – called PHYTOPIXAL – is based on a combination of indicators relating to the environmental vulnerability of the surface water environment (slope, soil type and distance to the stream) and the agricultural pressure (land use and practices of the farmers). The combination of these indicators for each pixel provides the contamination risk. The scoring of variables was implemented according knowledge in literature and of experts.

This method is used to target specific agricultural input transfer risks. The risk values are first calculated for each pixel. After this initial calculation, the data are then aggregated for decision makers, according to the most suitable levels of organisation. These data are based on an average value for the watershed areas.

In this paper we detail an application of the method to an area in the hills of Southwest France. We show the pesticide contamination risk by in areas with different sized watersheds, ranging from $2\,\text{km}^2$ to $7000\,\text{km}^2$, in which stream water is collected for consumption by humans and animals. The results were recently used by the regional water agency to determine the protection zoning for a large pumping station. Measures were then proposed to farmers with a view to improving their practices.

The method can be extrapolated to different other areas to preserve or restore the surface water.

1. Introduction

For several decades, powerful agricultural pressures on environments of varying vulnerability have led to the contamination of water bodies by a range of different pollutants: fertilisers, primarily nitrogen (Henin, 1980; Schröder et al., 2004; Perrin et al., 2008) and pesticides (Van der Werf, 1996; Barriuso, 2004) are among the best known. At the same time, society is demanding an ever more stringent protection of the environment. In response to this, public authorities have implemented numerous regulatory and incentive measures, some of which directly concern agricultural activities (CEE, 1991a,b, 1998).

In 1992, the European Economic Community implemented the first major reform of the European Agricultural Policy (Deneux and Emorine, 1998; Weyerbrock, 1998). The aim was to decrease the agricultural budget of ECC with a limitation of productions, and to increase the environmental protection (surface waters, groundwater, soils, and biodiversity). This new policy led to agri-environmental measures, with subsidies for farmers who decided to adopt environmental protection measures. The year 2000 saw the arrival of the European Framework Directive on Water, which requires all bodies of water to be in a good ecological condition by 2015. The Directive set out chemical and biological norms to be achieved by EU states (EC, 2000).

* Corresponding author. Tel.: +33 557890845.
E-mail address: francis.macary@irstea.fr (F. Macary).

Different assessments of these policies indicate that the sustainability of water management in rural areas can be improved. Access to decision-making tools at different spatial scales will allow a better analysis of risks, and help in the selection of those zones where certain measures should be prioritised. Changes in agricultural practices tend to take place at farm and land parcel level. This intermediate scale (small area) is useful for the implementation of actions by local groups (farmers within the same small watershed, local fishing associations, etc.). On the contrary, it is at the scale of water agency hydrological areas that different measures for water protection are defined. For instance, French policy makers recently decided to protect pumping zones for surface water by creating territorial action plans (TAP). A TAP includes measures in which participants (principally farmers, but also municipalities and inhabitants) are required to reduce the contaminants transferred into surface water. Some TAPs are concerned with the extension of culture successions on the same parcel of land (four or five different cultures, before returning to the first), reducing doses of fertilizers and pesticides, adoption of intermediate crops in winter bank protection, the Best Management Practices for tillage, etc. Some good environmental practices, like the introduction of vegetative filter strips along streams, can also be made more effective by the development of riparian zones, when possible. In these cases, managers give subsidies to farmers, and therefore need accurate information about potential risk zoning before drawing up their TAP.

Up until now, agri-environmental indicators and hydrological models used to assess the condition of water resources and the potential for contamination by pollutants (cf. Section 2.1), have only provided information at a single spatial level. These basic indicators were usually applied at the agricultural parcel scale and – more rarely – at the watershed level especially those of large dimension, where there are implications for environmental management (Devillers et al., 2004). These models cannot be applied to other scales, because data for the indicators are generally calculated at the parcel level. This kind of models tends to be applicable to best management practices in farming. However, for public managers and water agencies, the implementation of agri-environmental policies requires decision-making tools for larger scales (river basin, region, etc.) (Gardi, 2001).

To achieve this, the variables, geographical objects and basic indicators used, must be adapted to the transfer of spatial scales. In terms of definitions, we consider that the *vulnerability* of an environment (surface water for instance) represents the likelihood that it will be affected by pollutants. On the other hand, the *sensitivity* of the environment allows us to evaluate how this environment may respond to contamination.

The key environmental issues concern the conservation, preservation and rehabilitation of the various uses of a water resource. Indeed, different uses of aquatic environments may be affected by occasional and diffuse pollutions: abstractions for drinking water supplies, water resources for irrigation and recreational uses (fishing and bathing). The risk arises from the combination of these environmental issues, the vulnerability of aquatic environments and the different pressures being experienced.

The vulnerability of an environment plays an important role in the quality of surface water used by humans and animals. Pollution observed in rural watersheds mainly originates from agricultural inputs (pesticides, nitrogen fertilizers). Environmental risk assessment depends on the level of observation by the people involved and thus can be carried out at different spatial scales, according to the issue being considered: e.g. decision-making scales, agricultural practices or water contamination. Thus, a risk (CORPEN, 2003) allows an estimation of the possibility of changes to specific uses, such as the quality of water for consumption.

These different spatial entities may be superimposed and fitted together: nested watersheds when each entity is contained in another larger entity (a "Russian doll" approach), or become embedded, when only a part of the territory or spatial entity is included in another territory or entity. Particularly for agricultural management entities, e.g. farming parcels, farms and operational infrastructures, the same types of spatial entities very often fit together.

The problem of scale changes associated with agri-environmental risks is now a recurrent topic in sustainable development discussions; it is both a research concern and a prerequisite for supporting stakeholders in decision-making.

The aim of this paper is to present a new spatialised method, for assessing the contamination risk of surface waters by agricultural pesticide over interconnected areas of different sizes in different countries, implemented by combining different indicators. Our main objective is to provide a spatialised method using a GIS, which must be precise enough to obtain efficient results, and also to be suitable for environmental managers and stakeholders in their decision making. *Unlike hydrological models, we do not consider the assessment of contaminant flux, which need a resolution of complex equations, but a discrimination of risk areas by a suitable method for water and environmental managers.*

After a review of the different methods used for assessing water contamination risks, and for changing spatial organisational levels, we will explain the structure of the method that we developed (Section 2.2). We will then detail the test application, carried out in southwest France, where we used our method to, determine pesticide contamination risks for surface water destined for human and animal consumption.

2. Considering levels of spatial organisational in risk assessment: a review

2.1. Models and indicators used for characterising water contamination risks

Bio-physical models and agri-environmental indicators are generally used for assessing the contamination risk to water in watersheds (Tong and Chen, 2002). However, these approaches are specific to a single observation scale and they cannot be directly transferred to other scales.

Bio-physical models evaluate the influence of a factor and its modalities on the pollutants fluxes observed at the outlet of a river basin. Scenario testing is frequently used to provide answers in studies of complex phenomena. These models provide useful results for the estimation of pollutants flows, e.g. the EPIC model (Erosion Productivity Impact Calculator) (Williams, 1990), SWAT model (Soil & Water Assessment Tool) (Arnold et al., 1998), STICKS model (Brisson et al., 2003) and TNT2 model (Topography-based Nitrogen Transfers and Transformations) (Beaujouan et al., 2002). Wohlfahrt et al. (2010) implemented the MHYDAS model (distributed hydrological model for agrosystems) to carry out risk assessments for diffuse pollution in diagnostics as well as test-mitigation strategies in a small watershed. Ferrant et al. (2011) have recently developed the SWAT and TNT2 models for measuring nitrogen fluxes in South-West France. Boithias et al. (2011) and Lescot et al. (2011) used the SWAT model to assess pesticides fluxes in the same area. Lim and Engel (2003) refined the NAPRA (National Agricultural Pesticide Risk Analysis) model in the USA. Other models are currently under development for the same or similar purposes (Gascuel-Odoux et al., 2009a,b). These combine agronomical-hydrological models (which take into account agricultural practices at relevant spatio-temporal scales), the interaction with the physical environment, and the climate (Vazquez-Amabile and Engel, 2008). However, it is often difficult to set parameters and recover required data. Consequently, these models are more

useful as way of helping researchers to better understand physical processes than as a decision-making support tool for managers of public services.

Agri-environmental indicators are based on descriptive parameters that summarise the complexity of situations or processes (Bockstaller et al., 1997). They are used to monitor and assess the impacts of agriculture on the environment (Buczko and Kuchenbuch, 2010).

Most of them are built at the "farming parcel" level (Devillers et al., 2004), for example ADSCOR (Additive Scoring), EIQ (Environmental Impact Quotient), EPRIP (Environmental Potential Risk Indicator for Pesticides), GUS (Groundwater Ubiquity Score), I-Phy (Indicateur Phytosanitaire), PEI (Pesticide Environmental Index), or the Phosphorus Index (Bechmann et al., 2005), though some can be used at the 'farm' level: GUS, PEI, RMI (Risk Management Indicator). By combining these indicators with geographical information systems (GIS) – normally at the small watershed scale – it is possible to establish environmental risk zones for sensitive environments. GIS have been commonly used since the 1990s to determine the relationships between agricultural activities, including land use and water quality at the watershed scale (Srinivasan and Arnold, 1994; Xinhao and Yin, 1997; Aspinall and Pearson, 2000). They thus represent a decision-making support tool for the choice of measures to be implemented by managers of public services (Schröder et al., 2004; Alkan Olsson et al., 2009). Additionally, some indicators assess agricultural landscapes for water quality protection (Gascuel-Odoux et al., 2009a,b).

However, all of these indicators are built at a certain spatial organisational level and do not allow for changing of spatial scales. Hence, they generally cannot address the questions of environmental managers.

2.2. Methods for changing spatial organisational levels

Scale refers to the spatial dimensions at which entities, elements and processes can be observed and characterised (Dumanski et al., 1998). This conceptual space relates to study compartments and three levels are traditionally included: the global scale, the intermediate scale and the local scale.

The global scale concerns the environment and major systems (oceans, atmosphere and continents) for which the description grid is approximately one hundred kilometres.

The intermediate or "regional" scale corresponds to the regional engineering scale in particular, and allows the analysis of interactions between mankind and the environment. This scale takes into account the characteristic values relating to different processes, the geomorphological and hydrological organisation of areas at their best level of consistency (e.g. a watershed) and the level of anthropogenic activities. Therefore, agri-environmental issues are generally situated at this intermediate scale which varies from a few square kilometres to several thousand. This explains why different methods can be used.

The local scale concerns the processes themselves; these may be biological, such as plant growth, or physico-chemical, such as riverine transports of solute and particulate elements.

The *spatial scale* transmits the idea of continuity. In contrast, a *spatial organisation level* (SOL) corresponds to a spatial entity and the characteristic values of a process or of a phenomenon. This notion of organisational levels can simultaneously take into account the handling of ecological, agricultural and administrative processes. Four types of spatial organisational levels can be identified in the agri-environmental field:

- The level of eco-hydrological organisation (watershed, landscape unit);

- The level of socio-economic organisation: agricultural activity (farming parcel, farm, operational infrastructure);
- The local territory organisation with a collective identity (towns and other urban areas)
- The politico-administrative organisation or decision-making level for environmental managers (hydrographic area or basin, region).

In this paper, we will look at the eco-hydrological and politico-administrative organisation, as this is the level at which contamination risk assessment must be developed to aid environmental managers in their decision making.

2.2.1. Relationships between data sources and spatial scales

The relationships and interconnections between spatial entities without common geographical boundaries are sources of complexity in agri-environmental approaches (Dumanski et al., 1998; Dalgaard et al., 2003). For example, statistical data derived from the French agricultural census (Recensement Agricole) cannot be superimposed over a hydrographic division.

Specific sources of data (parcel surveys and farm surveys in addition to site, hydrological, statistical and remote sensing surveys) are obtained at distinct observation scales (Bock et al., 2005). The same applies to different environmental diagnosis methods (parcel reports, specific farm reports, indicators, spatial analyses, models).

Rather than remaining at a given level, a change of scale is the process of changing from one organisational level to the next. To do so, the variations in operation, schematisation, descriptors and the possible links between the consecutive levels must be specified (Puech et al., 2003). The development of GIS has facilitated the processing of these multi-source and multi-scalar databases (Balram et al., 2009).

The key aim is to have access to relevant information at the different spatial scales, while processing data which can aid decision-making for the managers of rural areas. Relationships exist between the study scales, data, methods used, the expected results and the data acquisition modes (Cotter et al., 2003; Dixon and Earls, 2009).

2.2.2. Reference Spatial Object

The complex interaction between the spatial organisational level (established by the decision-making level) and the spatial heterogeneity (to be measured or represented) is a major difficulty in the agri-environmental field and especially in hydrology. This difficulty is magnified if spatial heterogeneities need to be evaluated at multiple scales of decision-making. Wood et al. (1988) proposed the "representative elementary area" REA concept, or Reference Spatial Object (RSO), which takes into account that a phenomenon being studied corresponds to a working area with an associated scale. When there is a good match, the modelling or explanation of a phenomenon is both better and easier to understand. This method is proposed for defining the size of a watershed that is adapted to hydrological studies.

A Reference Spatial Object or discrimination unit can then be associated with each of these scales. The choice of an RSO is the result of a compromise between its appropriateness at the scale in question (optimal resolution), and its ability to accurately define the problems of diffuse pollution.

Changing from one scale to another causes major problems. For example, this occurs when the observations of agricultural pollution on basic parcels are related to the measurements of pollutant products at the outlets of major watersheds (Puech et al., 2003). This problem may be accentuated if we consider the temporal dimension (difficulty of comparing indicators of pressure and indicators of condition due to the retention and salting out of molecules which may persist over several decades).
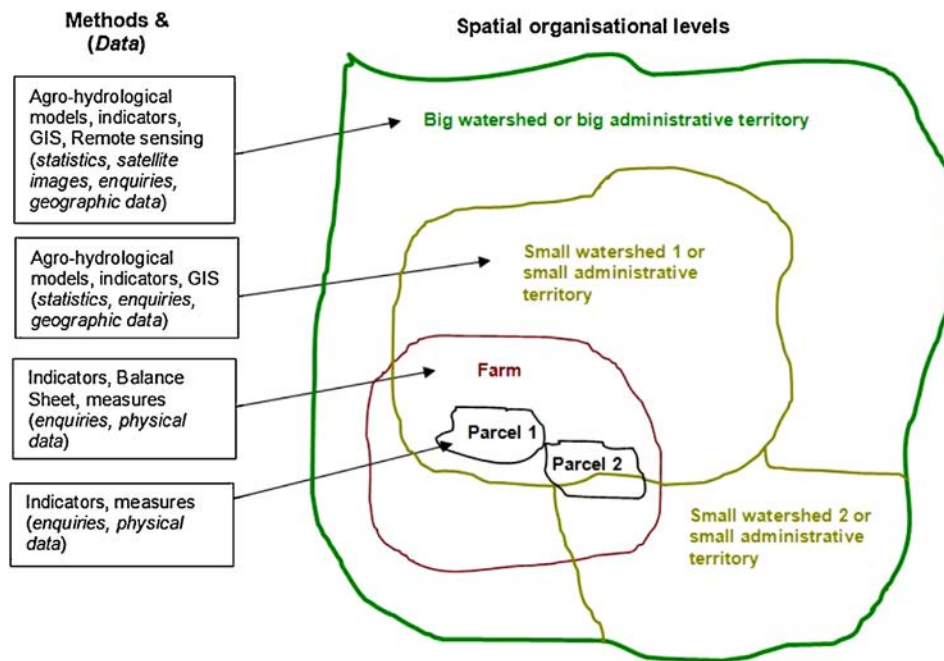
Fig. 1. Different spatial organisational levels, data and methods.

One way to resolve the problem of scale changing is to include several organisational levels when creating an indicator (Bock et al., 2005).

Indicators constitute an important element in the change from one scale to another and are becoming increasingly widespread in the handling of agri-environmental issues (Stein et al., 2001; Riley, 2001). They may have a single or multiple component(s). Riley (2001) emphasises the fact that in the agricultural and natural resource sector, the creation of a single-component indicator is the dominant approach. In contrast to the single-component indicator, multi-component indicators (Van Cauwenbergh et al., 2007) may be coefficients of variation or a complex function of these different components (Riley, 2001).

2.2.3. Methods for changing spatial scales

Different methods can be used according each context (Macary and Vernier, 2007a).

2.2.3.1. Specific indicators for each scale. The level of spatial organisation being studied determines the methods that have to be used. For each situation (Fig. 1), the response to the question of scale change is provided by a change of data and, consequently, of the environmental indicators specific to each spatial level (Riley, 2001; Stein et al., 2001). This is the main type of study carried out to date by the different authors. Indicators are built at different levels: parcels of land, farms, watersheds, and administrative territories (Macary et al., 2007).

2.2.3.2. Aggregative and disaggregative approaches. The description of phenomena within a spatial object often requires working at the previous or following organisational level. Aggregative and disaggregative methods often originate from these inter-scale approaches (Blöschl and Sivapalan, 1995).

Disaggregation is used to determine the behaviour of constituent components based on the behaviour of the whole entity (Riitters, 2005). Blöschl and Sivapalan (1995) describe disaggregation as being the transition from a "mean value" in a particular field to its detailed distribution within a section of this field.

Aggregation is used to make the transition from the constitutive components to the whole entity. However, in this case, the transposition of the processes identified at one scale offers no guarantees of validity at another scale. Furthermore, another difficulty relates to the interaction between several processes. While the representation of a single process may be envisaged, the inclusion of multiple processes is more complex.

Aggregation follows the same logic as an indicator which condenses raw information into synthetic information.

2.2.3.2.1. Method of direct aggregation at each RSO observed. The first method consists of aggregating the elementary data at each RSO observed.

For example, with agri-environmental information, scores qualifying simple indicators allocated to each parcel of land can be combined at the scale observed (each watershed level). This form of aggregation can be used to take into account different physical processes at the scales considered. The main disadvantage of this approach is the potential loss of information due to premature aggregation. The best solution for keeping as much of the information as possible is to choose a small RSO, calculate the indicators at this scale, and ultimately achieve aggregation at the organisation level selected by managers and actors.

Some authors have previously used this kind of aggregation, through a GIS process, of simulation units, assessed by the agro-hydrological model SWAP (soil-water-atmosphere-plant), for upscaling of water productivity, from fields to regions (Wesseling and Feddes, 2006).

2.2.3.2.2. Choice of a common element for the division of space at each scale (Aggregation and disaggregation). Another method considers a common RSO at the different scales for environmental managers and local participants. The adoption of the unit at the smallest possible resolution allows for the conservation of the most detailed information, regardless of the scale that is used, thus avoiding the premature aggregation of information.

Each element is associated with the value calculated, based on the information relating to the simple indicators. The aggregation, according to an arithmetical or statistical process, is then exclusively based on these elements for each of the scales determined in order to allow for the reading and interpretation of the results. The

accuracy of this spatialised method varies according to the level of spatial organisation. It is all the more important when there is a small area, given the available data and their details. This means that it is possible to supplement the information according to the data available at each organisational level.

Dalgaard et al. (2003) distinguished three aggregation procedures: the linear procedure, the non-linear procedure and the "hierarchical" procedure. The latter makes it easier to overcome the difficulties of interactions between processes.

The linear aggregation procedure is the most commonly known. It consists of the sum of its parts. In the non-linear aggregation procedure, the variable taken into account, can take different values, according to several thresholds. The aggregation must take into account the intrinsic characteristics of the fundamental units. The hierarchical procedure takes into account the intrinsic characteristics of each element of the system and also the relationships between these elements.

Disaggregative approaches depend largely on the data set available. The general method for the disaggregation of information is spatial interpolation. This method relies on tools directly characterising the form of spatial organisation and includes a large number of techniques. Some indicators are used to measure the shape of a spatial distribution. Here, the autocorrelation indices are the more traditional. Other methods of geostatistics use space in a more local way, such as variograms. These can identify breaks in the distribution of data or interpolation techniques and kriging (Ernoult et al., 2003).

2.2.3.2.3. Other aggregation processes, with multi-criteria approaches. Methods called Multi-Criteria Decision Aiding (MCDA) methods, or Multi-Criteria Decision Analysis methods, were developed in the 1970s (Roy, 1990). They have been used in several domains (Schärlig, 1996). Since the early 1980s, they have been tested with success in the field of environmental management issues (Simos, 1990). In most cases, the criteria aggregation process was based on a system of outranking (Maystre et al., 1994). Later, the MCDA was coupled with GIS, due to the spatial nature of environmental issues.

Malczewski (1999, 2006) carried out an important survey of literature with regard to GIS combined with MCDA and their many applications. Two main trends were developed. The first involved combining GIS software with MCDA methods, which makes best use of the performance of each system. Although very precise, this method involves many implementations between the two processes. Then, when these methods were applied on large territories, an integrated coupling with GIS software was developed, by addition in the GIS of a Multi-Criteria Evaluation (MCE) function (Eastman, 2001, 1993). The MCE take into account a fuzzy logic, and a criteria weighting (Saaty, 1990), like in classical methods: by this way, the GIS becomes GAS (Geographic Analysis System). Clarklabs of Clark University in USA developed the IDRISI software (Eastman et al., 1993). It was used in many environmental applications, like an assessment of land use in a large region (Paegelow and Olmedo-Camacho, 2005; Paegelow and Camacho-Olmedo, 2008).

3. Materials and methods

3.1. Study area and development context of the risk assessment method

In South western France, the Coteaux de Gascogne area is drained by 17 rivers. The five largest rivers in length and flow are from West to East: Baïse, Gers, Arrats, Gimone and Save. They are all left tributaries of the Garonne River, located between the Pyrenees Mountains and the Atlantic Ocean. All these rivers have their source in the Lannemezan plateau, and they are draining. A canal

between the Neste River and the sources of these Gascogne rivers was created in the second part of the 19th century to replenish these rivers with water during the dry period. Their watersheds cover several thousands of square kilometres. Over the last two decades (1985–2004), the mean annual rainfall was 691 mm with evapotranspiration of 819 mm.

The natural hydrological regime of these rivers is pluvial and the low flow period corresponds to summer (July to September) and the interannual variability of their module is very high (Probst and Tardy, 1985).

This geographical area (Fig. 2) covers 7000 km^2, which justifies the use of satellite images in order to obtain detailed coverage of the land surface. Data from the Corine Land Cover data base are not precise enough in terms of thematic resolution to establish potential risks of surface water contamination, as each kind of crop has to be considered, and not only arable lands as a whole.

Raw water for consumption by humans and animals is essentially extracted directly from these rivers by pumping stations (Fig. 2) and this is the main responsibility of water managers. Given the problem of relatively limited flows, contamination due to the transfer of fertilisers (mainly nitrogen) and pesticides of agricultural origin is magnified.

The effects of the agri-environmental measures to which farmers subscribed voluntarily after the new European agricultural policy, in 1992 (see Section 1) were very limited. Indeed, they were spread very thinly across the region and they made therefore little or no contribution to the required improvement on the pressures and their impacts on surface waters. This has prompted water agency managers, as in the Adour-Garonne river basins, to make special provisions for the protection of watershed areas. For example, the implementation of financial incentives had to be more precisely targeted, and were thus designed with the specific aim of reducing the pollution of river waters. Firstly, limits for the protection of water pumping areas had to be precisely defined, and there was a need to delineate high-risk water areas.

This called for the scientific development of a specific method to spatialise surface water risks, easier to use than classical hydrological models and adapted to different spatial scales. In this context, we carried out research into a method of risk determination, at different spatial scales and applicable to other regions in which surface water is exposed to different contaminants.

In this area, we considered nested watersheds as different spatial organisational levels (SOLs). For example, the largest watershed (Save, 1150 km^2) is the result of interconnections of secondary watersheds drained by tributaries (i.e. Boulouze river basin, 70 km^2), which are themselves interconnections of elementary watersheds drained by sub-tributaries (i.e. Montoussé catchment, 6 km^2).

3.2. Implementation of a spatial cognitive method

3.2.1. Overall structure of the method and its different SOLs

Our generic spatialised pixel-based method (called PIXAL method) is based on a combination of spatialised agri-environmental indicators, calculated at the pixel level of remote sensing. The application in this paper focuses on phytosanitary issues, and the method is therefore called PHYTOPIXAL.

PHYTOPIXAL takes into account factors of surface water vulnerability (Section 3.2) and agricultural pressure (Section 3.2.3). In a first step, each factor provides a basis indicator, usable to discuss with stakeholders. Then, these indicators are combined to obtain risk values, calculated for each pixel (30 m × 30 m).

In a second step, risk values are aggregated to determine risk zones, according to the level of organisation and management chosen by the public management bodies and stakeholders.
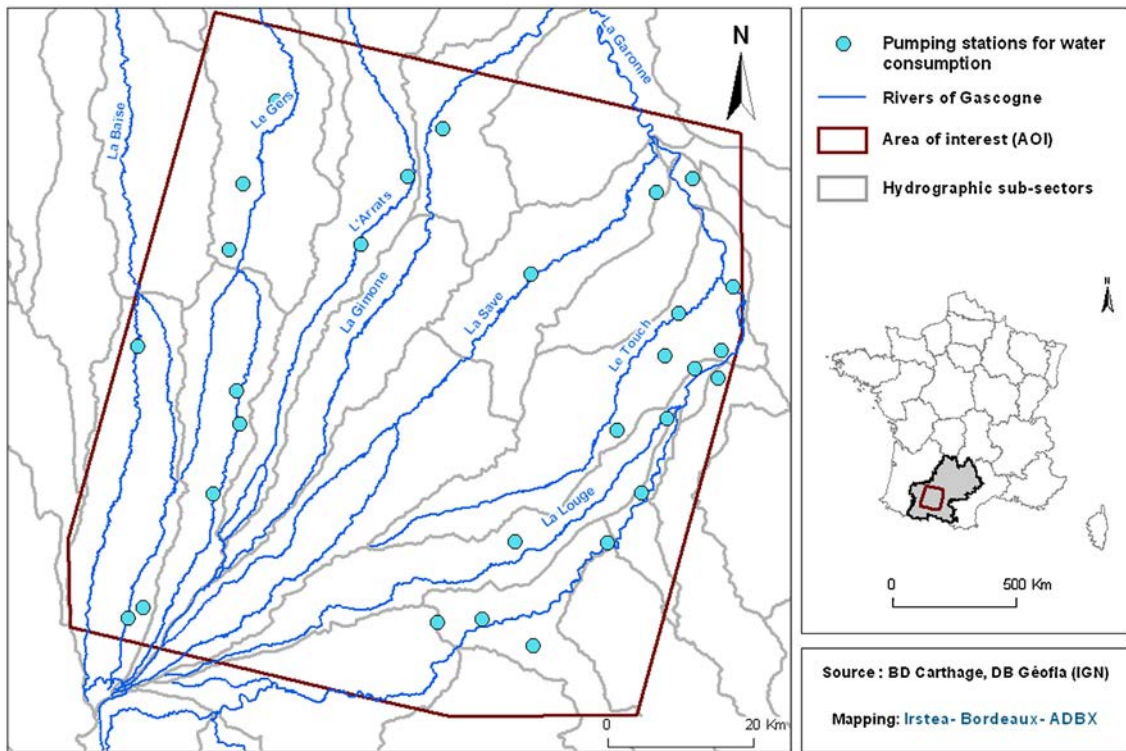
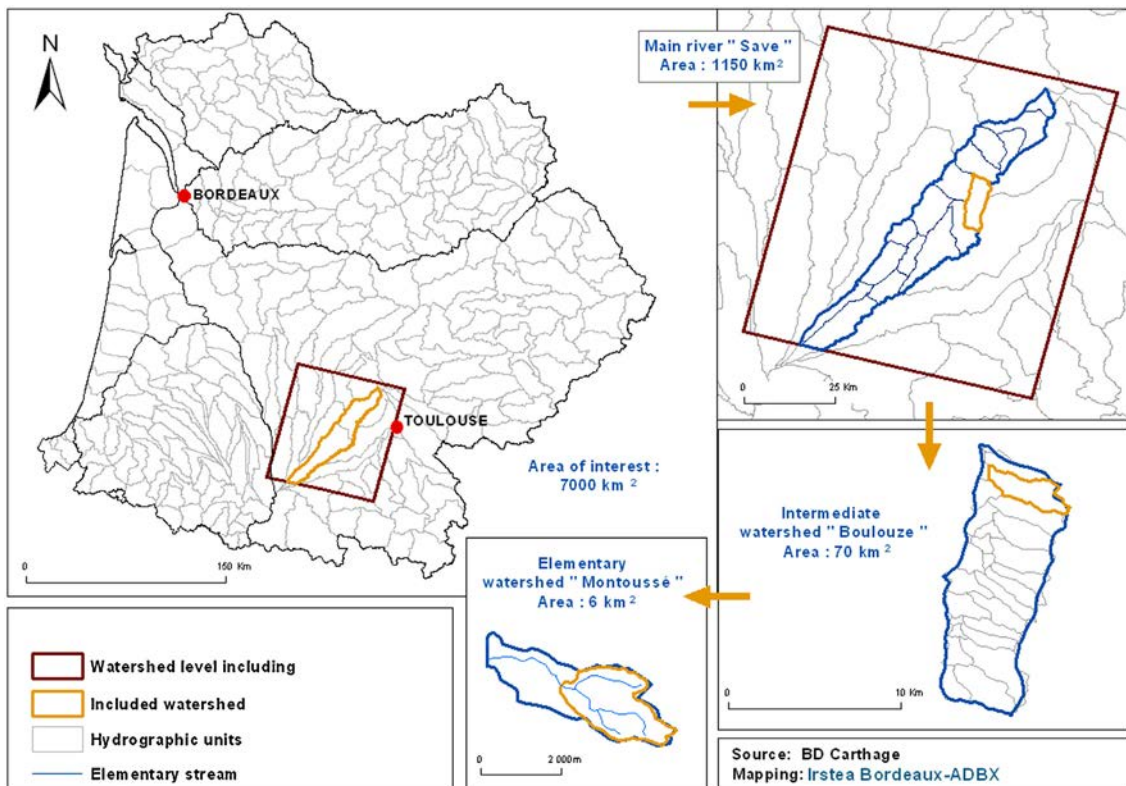**Fig. 2.** Location of the study area in Southwest France.

Fig. 3. Example of different spatial organisational levels in the "Coteaux de Gascogne" area.
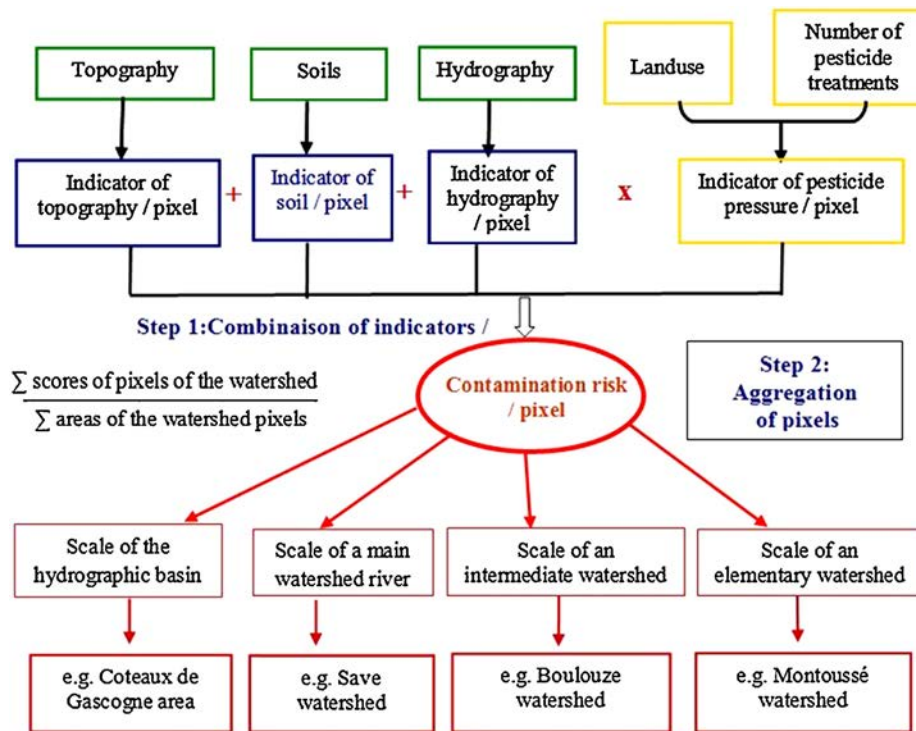
Fig. 4. Overall structure of the PHYTOPIXAL spatialised method and its different scales of results.

From these data, the contamination risk of surface waters by pollutants was calculated at different SOLs (Fig. 3), and expressed from the scale of the small elementary agricultural watershed to that of the large surrounding watershed. The general flowchart of this method appears in Fig. 4.

3.2.2. Creation of surface water vulnerability indicators

The factors which contribute to surface water vulnerability are mainly related to the nature of the soil, the geological substratum, the slope, and the connectivity with the body of water in question. Some of these were discarded because they were not relevant for the characterisation of different scales of interest, or did not provide data sufficiently accurate and discriminating in the study area (i.e. geological substratum). The parameters retained were topography, drainage and soil. This information fully satisfies the need to incorporate the change of scale. Data were gathered over the full area at the smallest scale, allowing for a more precise description.

All variables were scored (Devillers et al., 2004) prior to the calculation of the indicators. From the range of variation of each factor, classes of values were determined according to previous knowledge (based upon scientific literature, agronomists' expert opinion, and our experience in this area) regarding their contribution to contamination risks. That explains why we considered this model as a cognitive model (Le Bissonnais et al., 1998). We attributed the highest score where there was the highest level of risk, and identical weighting was used for all scores.

Table 1 summarises the scores of the three vulnerability indicators.

3.2.2.1. Slope.
Slope is an essential factor in the transfer of pollutants like pesticides (Beaujouan et al., 2002; Barriuso, 2004), influences surface run-off, and has a direct effect on transfer times from soils to water. Furthermore, it has direct consequences on physical erosion processes and facilitates the transportation of active substances adsorbed onto suspended matters particularly during the flood events (Taghavi et al., 2011). A Digital Elevation Model (DEM) from the French National Mapping Agency at a resolution of 25 m was acquired to cover the study area. Extrapolation to a resolution of 30 m allowed the same spatial resolution as that of a satellite image.

The thresholds were determined according the specific literature about erosion in France (Le Bissonnais et al., 1998). They were manually calculated, based on the frequency histogram of the slope values, at the basic pixel level. As the extreme values were limited in number, we incorporated them into the major classes. Five slope classes adapted for surface runoff to calculate scores for this factor, taking into account the main characteristics of each slope class: <3%; 3–7%; 7–12%; 12–25%; >25%. In this study area, crops are mainly grown in parcels with less than a 12% gradient. Beyond this value, we found mainly grasslands. In this study, we used scores from 1 to 5 (Table 1) corresponding to the previous slopes classes

Table 1
Scores of vulnerability indicators regarding slopes classes, soil nature, distance to hydrographic network.

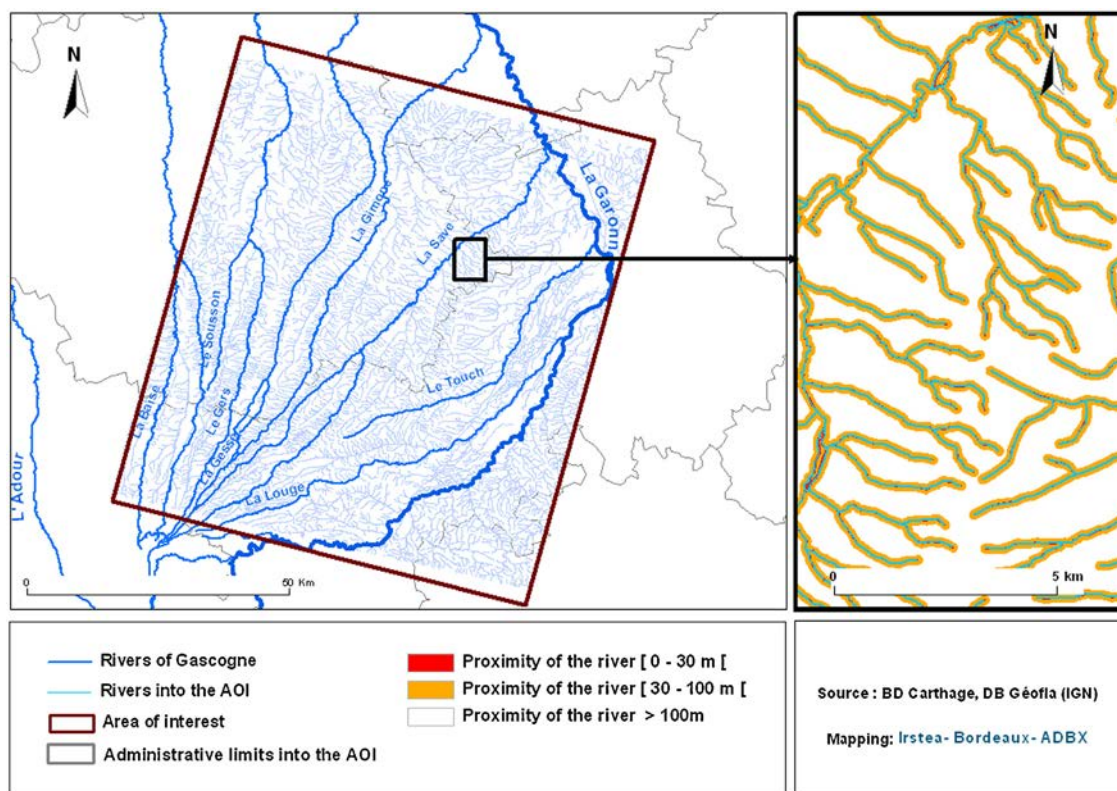| Vulnerability indicators	Score/pixel
Slopes classes	
>25	5
[12–25[	4
[7–12[	3
[3–7[	2
[0–3[	1
Soil nature	
Rendosols and thin calcosols	4
Thick calcosols	3
Luvisols and cambisols	2
Fluviosols	1
Distance to hydrographic network	
>100 m	5
30–100 m	3
<30 m	1

Fig. 5. Distance to watercourses.

(1 for slopes <3%, 5 for slopes >25%). These scores must be adapted to different geographical contexts, in other areas and countries.

3.2.2.2. Distance to watercourses. The contamination of surface waters by pollutants is highly dependent on the distance between the application site and the relevant body of water. Two parameters were considered when developing the indicator of distance to the drainage network: the exact definition of the active network and the choice of "distance" adopted. The processing of the database for the drainage network consisted first of extracting the permanent drainage sections. We then divided up the resulting layer, based on the superficial area of the satellite image. We created two buffers, situated at distances of 30 and 100 m around the drainage network in order to obtain three distinct zones (Fig. 5).

The threshold of 30 m was based on the spatial resolution of the pixel. The 100 m threshold was based on local expert opinion, taking into account the very tormented topography in the field, especially in this hilly area: this was a first test of threshold implementation. We allocated a score to each of the zones (Table 1) after the merging and rasterisation of the extracted layer. This score simply shows the increased risk with proximity to the network in 3 distance classes: <30 m (score 5); from 30 to 100 m (score 3); >100 m (score 1). These values were assigned to account for the importance of this criterion, regarding other studies (Aurousseau et al., 1998). The context (connectivity) of this region required an adaptation of the scores depending on the spatial scale considered. They can be easily changed according each physical context.

3.2.2.3. Soil types. Soil characteristics play a highly significant role in the retention time of residues in the upper soil horizon to the stream waters. Humus content and nature of organic–mineral complexes can play very important roles in the retention of molecules.

Throughout the Oligocene and Miocene periods, this area of the Coteaux de Gascogne received sandy, clay and calcareous sediments derived from the erosion of the Pyrenees Mountains (Revel and Guiresse, 1995). The heterogeneous materials produced a thick detrital formation, called molasse, during the Miocene. From the Quaternary period onwards, the rivers became channelized, cutting broad valleys in the molassic deposits and leaving terraces of coarse alluvial sediments. The substratum of the different watersheds consists of impervious Miocene molassic deposits.

A weak erosion has allowed the development of calcic Luvisols on the tertiary substratum and local Rendosols (FAO, 2006) on the hard calcareous sandstone beds. A moderate erosion affected soil calcic Cambisols on hillsides with very gentle slopes (less than 12%). Calcic soils (Calcisols) are dominated by a clay content ranging from 40% to 50% and called locally "Terreforts": they are largely represented on the hills and because of their impermeability, they greatly contribute to surface runoff. Non-calcic soils (Luvisols) are silty (50–60%) and locally called "Boulbènes" (Revel and Guiresse, 1995).

In this area, 1/50,000 scale maps were drawn by pedologists from the CACG (Compagnie d'Aménagement des Coteaux de Gascogne) in the 1960s and have since been transposed to 1/80,000. This soil map was digitised for the Save watershed only (Cemagref, 2006), limiting the use of this information to this watershed. It revealed a significant but varying, soil depth in the river Save area, which could influence surface runoff. The soil types were grouped into four main categories, and scored according to their importance with regards to the transfer of pollutants by surface runoff (Table 1). This was based on the studies carried out by pedologists (Revel and Guiresse, 1995) and professional technicians belonging to a Regional Action Group against Water Pollution by Pesticide Products (GRAMIP). Soil types are ranked from the lowest to the highest potential surface runoff process below:

○ Fluviosols ("*Alluvions*"): The proximity to the superficial water table leads to direct transfer towards it and therefore an absence of transformation/degradation in the environment.

○ Luvisols and cambisols ("*Boulbènes*"): Permeable soils which facilitate infiltrations, but favour the formation of superficial crusting, which is conducive to surface runoff.

○ Thick calcosols ("*Terrefort épais*"), soils >40 cm: Impermeable, clayey soils; runoff is predominantly hypodermic. These soils are composed of hydromorphic sandy pockets which slow down the transfer of products to the water table.

○ Rendosols and thin calcosols ("*Terrefort mince*"), soils <40 cm: They favour hortonian (surface) runoff which occurs when rain intensity exceeds the infiltration capacity of the soil.

3.2.3. Composition of the agricultural pesticide pressure indicator
3.2.3.1. Land use.
Land use within a watershed influences runoff, hydrological regime, and stream water quality. Land cover by crops can be used to assess the pesticide pressures exerted by agricultural additives. The classification of satellite images constitutes the basic information layer of the method.

The Corine Land Cover database is not accurate enough to differentiate the crops types on farmland, thus impeding the precise determination of risk levels. Because of this, we carried out a supervised classification from satellite images using Erdas Imagine software®. Annual land cover is given by the classification of a series of four Landsat 5 TM satellite images, taken on multiple relevant dates, to characterise each type of plant per season, with the lowest rate of clouds: 0–20 max; 2004/12/01; 2005/05/26; 2005/07/13; 2005/08/30.

The area of these images covers the Coteaux de Gascogne area, at a spatial resolution of 30 m × 30 m.

Classification in supervised mode, i.e. under the direct control of the operator, locates the crops in a sufficiently accurate and exhaustive manner. The samples were grouped into ten categories of major crops that consume agricultural additives: wheat, barley, rapeseed, sunflower, corn, sorghum, peas and beans, soybean, grasslands and fallow. Classification also differentiates other land use: fallow, hardwood, softwood, water, soils and buildings. We applied the maximum likelihood rule to the classification algorithm: each pixel was allocated to the class with the highest mean probability of all of the batches.

3.2.3.2. The agricultural pressure indicator for pesticides.
The level of pesticide pressure exerted by agricultural activities can be derived from land use for the studied year, as each crop can be associated with a mean number of pesticide treatments. Given the large number of pesticide substances and various commercial formulations, we attributed a mean number of treatments to each crop (herbicides, fungicides and insecticides) according to data provided by the French Regional Department for Plant Protection (SRPV). The treatment of seeds, something which remains very localised, was not considered here.

Therefore, the pesticide pressure indicator resulted from the crossing of the classified image of land use with the number of pesticide treatments. To easily implement the scores in the GIS, these values were multiplied by two, in order to avoid decimals.

3.2.4. Determination of the contamination risk (CRk)
We performed a spatial analysis in order to optimise the discrimination of pesticide pressure areas at a regional scale, according to a division into elementary watersheds or hydrographic areas: levels of decision making for managers and stakeholders.

The layers of each single indicator were rasterised by defining a common resolution cell size of 30 m × 30 m. The contamination risk was then obtained by cross-referencing them.

Environmental vulnerability indicators were added with an identical weight for each factor. Based on current knowledge, rising indicator values show an increasing risk. Next, the contamination risk was obtained by multiplying the pressure indicator by the environmental vulnerability indicators (Eq. (1)). This allows the calculation of a zero risk in areas in which no treatment has been applied.

3.2.4.1. Calculation of the contamination risk for each pixel.
The following formula was adopted for the spatial cognitive model:

$$CRk = [SL + D + So] \times [Ap] \qquad (1)$$

where CRk, contamination risk for each crop, SL, values for the slope level, D, values for the distance from each pixel to the stream, So, values for the soil type, Ap, agricultural pressure (number of pesticide treatments by crop).

We used five risk categories, which allowed for greater discrimination at the scale of this region. Analysts and decision makers generally use this number of categories in risk studies.

We maintained discretisation according to Jenks (Lepinard, 2008) natural thresholds, thus obtaining the most "homogeneous" categories of possible values. The thresholds varied according to the number of discriminated units. The level of contamination risk allocated to a given unit (individual statistic) depended on the performances of the other units. These units are compared, with measurements always performed using the same indicator.

3.2.4.2. Aggregation of information according to the investigation scale.
In order to identify priority areas for action, the information obtained may be subsequently aggregated. According to the different solutions described in Section 2.2.3, we chose the pixel or grid of a satellite image as our common RSO (Reference Spatial Object). We aggregated them using a linear procedure (Section 2.2.3.2.2), at the different spatial organisational levels, in order to be usable by environmental public managers and stakeholders (farmers and their advisers). To do this, we used spatial statistics tools (Spatial Analyst of ArcGIS software®, which does not include MCE function) according to the following formula (Eq. (2))

$$\mathbf{CRk}_{watershed} = \frac{\sum \text{scores for pesticide risk in pixels}}{\sum \text{surface area in pixels}} \qquad (2)$$

This is an area-weighted average of risk values calculated at the chosen scale.

4. Results

4.1. A conceptual model to generalise the contamination risk assessment

The method developed here aims at assessing surface water contamination risks, at different organisational levels. It provides valuable information at scales appropriate for environmental managers and stakeholders, in order to support decision making. It is suitable in different areas and countries where the main water transfers are superficial. This method is unapplicable in its present forms, for example in karstic environments. Fig. 6 shows the conceptual model of the generic PIXAL method.

Our assessment method was used to determine the level of pesticide contamination risk by in the study area.

4.2. Result of the classification of the remote sensing images and pesticide pressure indicator

Results of the land use and consequently the pesticide pressure appear in Fig. 7 and Table 2.
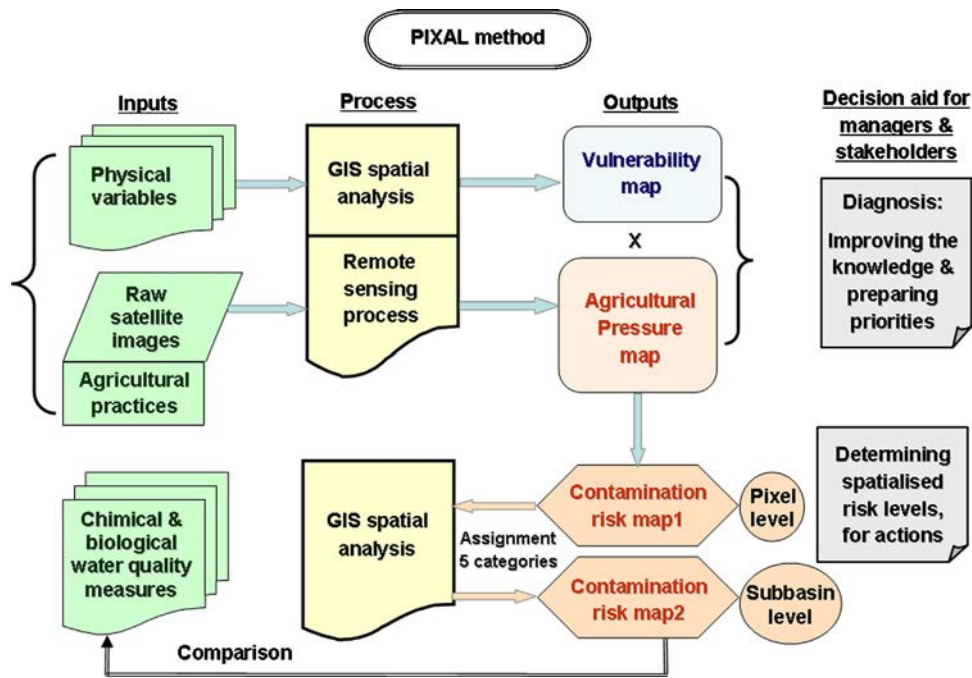
Fig. 6. Conceptual model of the generic PIXAL method.

In this area of 7000 km^2: water, buildings and woods represent 15%, and farmland use 85% (6000 km^2). Four categories account for 83% of farmland use: Grasslands (35%) essentially in the piedmont of Pyrenees, sunflower (19%) and wheat (18%) in north part, where agriculture is intensive; and fallow (12%) in all the area.

The overall quality of the classification was given by the mean percentage of correctly classified pixels. This corresponds to a measurement of the separability of the categories according to various channels. The Kappa coefficient (Congalton, 1991, cited by Ducrot, 2005) estimates the accuracy of a classification, which takes into account errors in rows and columns. We obtained a mean percentage of correctly classified pixels of 86% and a Kappa coefficient of 0.83 for the 2005 land use. This allowed us to accurately employ the land use coverage obtained. The Kappa coefficient is 0.92 for the wheat, 0.77 for the sunflower, and 0.87 for grasslands and fallow.

4.3. Level of pesticide contamination risk in surface water

We obtained a surface water contamination risk for each pixel. But this mosaic of 7,653,255 pixels cannot be easily interpreted by
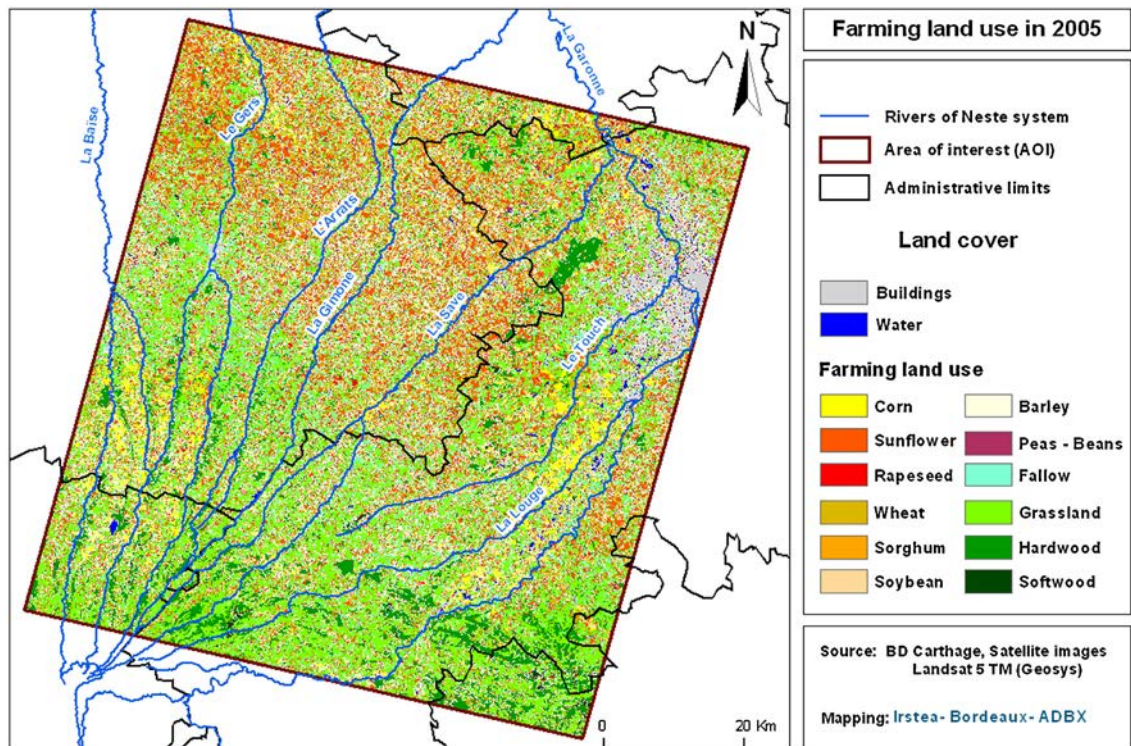

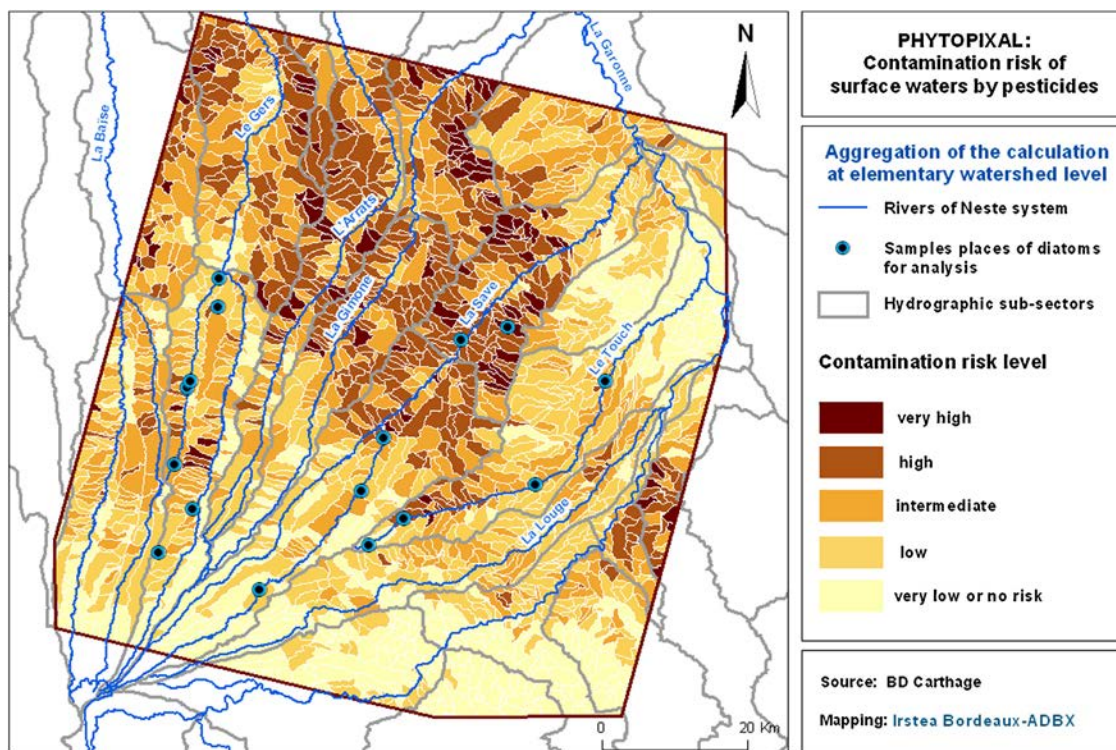**Fig. 7.** Land use in the study area.

Fig. 8. Estimated risk at the scale of the elementary watershed.

environmental managers. It shows the level of detail of the calculation, but cannot be used for agri-environmental decision making by itself. This is the level of intervention on the farming parcels (farmers and advisers). Then, we aggregated the pixels at the scale of elementary and intermediate watersheds.

4.3.1. Aggregation n°1

The pixels were aggregated first to elementary watersheds, and then to hydrological areas. Consequently, Table 3 and Fig. 8 represent the contamination risk (after aggregation) at the scale of the 2871 elementary watersheds, covering several square kilometres

(several km^2). This is primarily the local decision-making level for agri-environmental water preservation measures.

High and very high contamination risks (27% of the surface) are located in the areas where agricultural pressure is at its highest (Fig. 6), in contrast to areas where it is lower (59% of surface), including grasslands or crops with few pesticides, like in the Piedmont of Pyrenees mountains (south of the map). In Piedmont, water vulnerability is higher than downstream, mainly due to slopes and small valleys which provide short distances from land parcels to streams. The land cover is dominated by grasslands and forests without direct pesticide application. A few cereals are cultivated, on small plateaux.

4.3.2. Aggregation n°2

Table 3 and Fig. 9 represent the risk value at the scale of the 125 intermediate watersheds (tens of km^2). This is the decision-making scale for environmental managers in France. At this level, the aggregation of pixels is easier to read for managers. On the other hand, the information offered is of a more general nature and suited for the implementation of environmental protective measures.

The second aggregation provides an increasing number of risk assessments for high and very high levels and at the opposite, a decreasing of the low rates.

The number of Reference Spatial Objects between elementary watersheds and hydrological areas is divided by 23.

In this study area, the managers of the French regional water agency (Adour-Garonne) at the scale of the large watershed covered
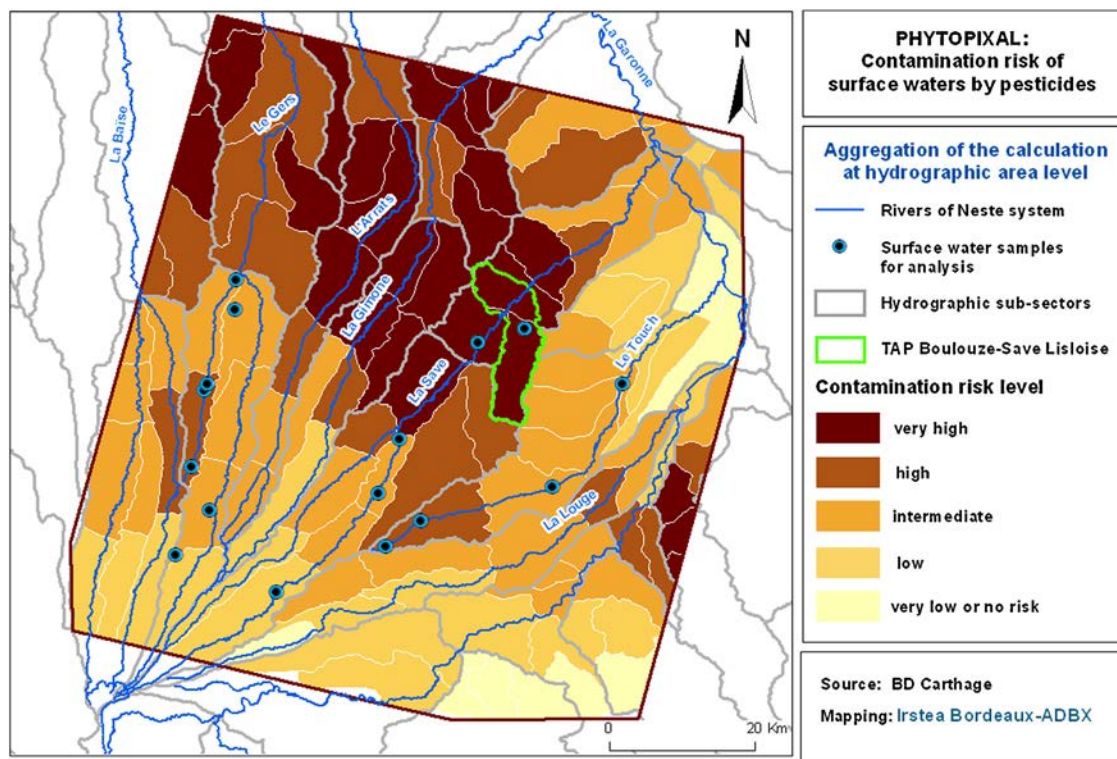
Table 2
Farming land use and scores of pesticide pressure indicator.

Pesticide pressure by land use type	Farming land use (km^2 and %)		Score/pixel
Rapeseed	161	2.7	12
Peas Beans	79	1.3	11
Wheat	1078	18.0	8
Corn	430	7.2	7
Barley	188	3.1	6
Soybean	101	1.7	5
Sunflower	1153	19.2	4
Sorghum	39	0.7	2
Grasslands	2075	34.6	0
Fallow	696	11.6	0
TOTAL	6000	100.0	

Table 3
Surface corresponding to the levels of contamination risks, depending on aggregation scale: (i) 2871 elementary watersheds and (ii) 125 hydrologic areas.

Risk level	Elementary watershed % total surface		Hydrologic area % total surface	
Very low	28.5	58.6	12.5	31.2
Low	30.1		18.7	
Intermediate	24.3	24.3	25.2	25.2
High	14.5	27.1	32.6	43.6
Very high	2.6		11.0	

Fig. 9. Estimated risk at the scale of the intermediate watersheds.

by this study, decided to choose a protective boundary for the zoning of an abstraction point for drinking water supplying a population of 20,000 inhabitants (the town of L'Isle Jourdain). The zoning of the Territorial Action Plan (TAP) of Boulouze-Save Lisloise was established on the basis of this contamination risk map for an intermediate watershed of 112 km^2 (Fig. 9). The TAP offered the opportunity for farmers in this area to sign up for specific agri-environmental measures, supported by French water agency subsidies from 2008 to 2012. Other Regional Action Programmes have been created under the same conditions in order to re-establish or maintain the surface water quality for human and animal consumption.

This method constitutes a step forward in the research into changes of scale and the evaluation of pollution in a large river basin. It can therefore act as a useful decision-making support tool for managers of public services in different countries. Then, we are applying it with success, in other watersheds of different size (>1000 km^2 and <100 km^2) in Spain and Portugal within the framework of a European Project, Aguaflash (see http://www.aguaflash-sudoe.eu/ for details Macary et al., 2011).

5. Discussion

5.1. Applicability of the PHYTOPIXAL method

The PHYTOPIXAL method can be easily and quickly implemented by environmental managers in any country, by adapting scores to each local context. However, this method has been designed predominantly for surface water contamination risks with surface runoff fluxes, and no important infiltration, like in karstic areas. While spatial scales for aggregation depend on a countries' size, this methodology is suitable to all the main farming countries in Europe, and even other regions with similar characteristics, in different continents and indeed outside Europe, in regions with similar characteristics Each aggregation level of risk results

will, of course, depend on the size of countries and regions, but also on the geographic level of public policies application.

The Reference Spatial Object (grid of 10 m, 20 m, and 30 m) can give very precise results for water vulnerability, agricultural pressure, and contamination risks. Each step of the assessment scale, with the corresponding indicators, leads to a greater understanding of agri-environmental processes, and equips decision makers to make the most informed choices possible. Aggregation of the RSOs at appropriate decision levels can help manage priorities in complex areas.

This method can be used to compare individual watersheds and rank them according to their stream water contamination risk. The maps acquired using this method help to define the priority areas for action and facilitate the dialogue between stakeholders by giving them common methods and documentation. In addition, these maps allow for the identification of target areas at different levels of organisation, which can then be studied more specifically (hydrological measurements, water quality measurements, specific monitoring of inputs of pesticides inputs, cultivation methods), or of priority actions in the fight against non-point pollution. They contribute to a risk diagnosis of pollutant transfers into surface waters, and help in identifying the best remedial and protective action. These benefits have been emphasised by different authors (Misra et al., 1996; Patty et al., 1997; Mérot et al., 1999; Schmitt et al., 1999; Gouy and Gril, 2001; Carluer et al., 2009).

5.2. Comparison with other methods

Different methods based on a combination of variables are generally used at the farming parcel level or at small watershed level but rarely on a large territory with the capacity to aggregate risk assessment at the different organisational levels used by stakeholders and environmental managers.

Nevertheless, some authors (Eastman et al., 1993; Jiang and Eastman, 2000; Paegelow and Olmedo-Camacho, 2005) used the

GIS' integrated multi criteria evaluation function to analyse large territories and to assess a prospective land cover (Section 2.2.3.2.3).

The use of software like IDRISI – a GIS which incorporates a multi-criteria evaluation function – could therefore be useable for an alternative application for assessing contamination risks.

Agro-hydrological modelling is generally used to improve the knowledge process by researchers (Arnold et al., 1998; Ferrant et al., 2011), but require a lot of data, which are often unavailable in a large area. Their implementation is not easy and time-consuming due to the complexity of the software. Moreover, they provide results within their pre-programmed scales, and do not adapt the data to suit the needs of environmental managers. Some of them, like STICKS (Brisson et al., 2003), TNT2 (Beaujouan et al., 2002), provide results at farming parcel level, which is useful for farmers, and in understanding hydrological transfers in small watersheds. Other programs, like SWAT (Ferrant et al., 2011) deliver results based on HRUs (Hydrological Response Units), which represent hydrographic areas of watersheds, without any connection with management Reference Spatial Objects. They are also used by researchers, but seldom by environmental managers, who prefer methods that can help them make fast and effective decisions.

5.3. Validation of the PHYTOPIXAL method

Decision-making support tools are intended to offer scientifically validated information where possible. They are designed to conform to a rigorous approach and to be used to support managers who often lack the resources and tools for managing situations with different levels of spatial organisation.

Like most methods for determining spatial risk, it is always difficult to validate the results produced on a large scale, where the institutional survey networks for pesticide contents are hugely inadequate. Finally, comparisons with the results obtained by other research projects cannot be achieved, because the goals and methods used for creating the indicators are divergent (spatial divisions, pressure maps, etc.). Nevertheless, a validation method, or at the very least a means of comparing the results obtained with those of bio-physical models, could be worth exploring, but differences in scales cannot really provide a validation, or a relevant comparison of the results.

With this in mind, we compared risk zonings at different organisational levels with biological indicators, which are take into account the impacts of pollution in aquatic environments. As pesticide contamination events are very transient and difficult to catch by spot samplings, we compared PHYTOPIXAL results to biological data that are expected to inform on their "exposure history". Comparisons made to determine diatom (fixed microalgae) responses to pesticide pollution led to encouraging results (Morin et al., 2009). We used a dataset of about 439 diatom samples, collected from artificial substrates (immersed in the waters under comparable hydrology and light conditions) at 16 sites in the Coteaux de Gascogne area (Figs. 8 and 9) in spring 2005 and 2006. Biological samplings were performed on different occasions (sampling dates) to characterise the diatom communities that settled in the streams during the period covered by the present PHYTOPIXAL calculations. At these sites, PHYTOPIXAL values ranged from low to very high, with 31 samples (7%) in low, 214 (48.7%) in intermediate, 147 (33.5%) in high and 47 (10.8%) in very high potential risk of contamination (no diatom sample in the very low PHYTOPIXAL level). From this biological data, we calculated an index of general alteration, the IPS (Indice de Polluosensibilité Spécifique, Coste in Cemagref, 1982) and species diversity (Shannon and Weaver, 1949). Anovas (Fig. 10) were performed on the dataset using STATISTICA v.5.1 (StatSoft, France). IPS values differentiated two sets of data: samples collected from sites with low contamination risk ($n = 31$) had good biological
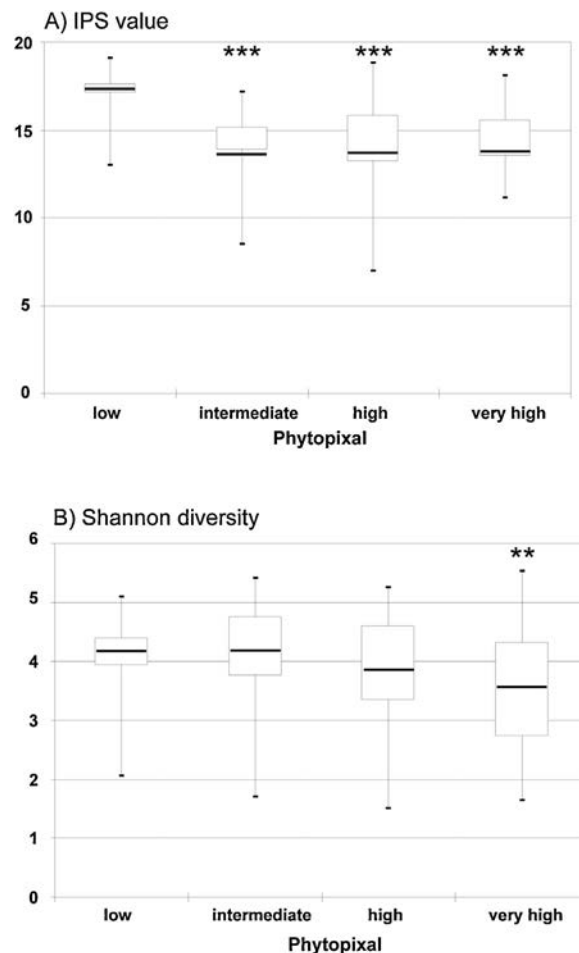

Fig. 10. Distribution of (A) IPS values and (B) Shannon diversity indices in the 4 Phytopixal categories tested. Values are minimum – lower quartile – average – upper quartile – maximum. Statistically significant differences after ANOVA: **$p < 0.01$, ***$p < 0.001$.

quality (as shown by high index values, IPS = $17.2 \pm 0.2/20$), whereas those located in areas with higher pollution risk had significantly lower index values (IPS = $14.5 \pm 0.1/20$, $n = 408$).

Specific diversity, which is known to decrease with pesticide pollution (e.g. Seguin et al., 2001; Morin et al., 2009), was fairly high in sites with low or intermediate pollution risk (div = 4.13 ± 0.05, $n = 245$) and dramatically decreased in the most contaminated locations (div = 3.53 ± 0.14, $n = 47$). Although rough, this comparison between risk zoning and real field observations constitutes a first step towards the validation of the PHYTOPIXAL method.

5.4. Limitations and potential improvements of the method

5.4.1. Determination of pesticide pressure

It would have been especially useful to have differentiated between the large families of pesticide molecules and their mixtures in commercial formulas, given their different degrees of harmfulness. In the same way, unidentified inert ingredients are often forgotten in commercial preparations, whereas they can have long-lasting toxicological consequences (Surgan et al., 2010).

However, at a scale of several thousand km², this degree of accuracy cannot be achieved for a method destined for use by managers, because it would be too difficult to integrate all the active ingredients.

5.4.2. Incompatibility of the scales of analysis and the water contamination process with the availability of data

Certain data are not available at the scales incorporating variables essential for the analysis of diffuse pollution risks from agricultural origin concerning superficial water. For example, some data which are useful for characterising environmental vulnerability are not available or are unsuitable for discriminating space (geology or soil at a sufficiently accurate scale). Likewise, some of the information required for calculating pressure remains imprecise. Therefore, agricultural production data were taken from local surveys and the results were then extrapolated to cover the entire study area.

The performance of field surveys in order to obtain this data at the level of a large watershed would be too consuming in terms of human resources. Applied to a large area, our method does not take into account local physical points such as grassy strips situated alongside watercourses. It would be worth exploring the possibility of acquiring high-resolution satellite images (e.g. Ikonos) over a restricted test area in any future research. Furthermore, with regard to agricultural practices, each pixel for a given crop is processed in the same way over the whole area, according to the hypothesis that practices for each crop, are homogenous in this area.

5.4.3. Transfer of information from one scale to another: limitations of the aggregative approach

We have previously seen (Cf. Section 2.2.3) that the aggregative approach reduces the performance of functional relationships. This is due to the arrival of new dominant processes which obscure the impact of local knowledge or measures, which therefore no longer have a numerical significance on the result. According to Puech et al. (2003), the problem concerns the allocation of a numerical value to a grid cell and the relationship between the different grids. On the one hand, fluxes of water or matter are associated with geometrically shaped cells which are often square shaped. On the other hand, the spatial aspects (description of cells) and temporal considerations (link between the cells) are confused. "The aggregation is often rendered complex, or indeed impossible, due to non-linearities and local heterogeneities exacerbated by a change of several organisational levels, from m^2 to watershed of 100 km^2, i.e. jumps of several orders of magnitude. When several levels are skipped, there is no longer a direct link between cause and effect." (Puech et al., 2003).

This approach would probably benefit from further development, especially in terms of scale change; nevertheless we managed here to aggregate vulnerability and pressure information at the different organisational levels in question. The resulting estimation of pesticide contamination risk in this study area was used by the regional water agency, and also tested in Spanish and Portuguese watersheds (Macary et al., 2011).

5.4.4. Subjectivity of zoning criteria and choices of mapping representation

During the modelling phase, we allocated scores to the indicators and opted for a discretisation method to produce final maps showing potential contamination risks. These choices were not made randomly, but were mainly based on expert opinions and as such, remain open to question. The most important point concerns the quest for operating thresholds. The difficulty resides in identifying those that appear to be variable within the same area according to the context and the variables or processes studied. Scored indicators offers benefits like their modularity and the fact that they are easy to understand by users (Devillers et al., 2004). There are advantages and disadvantages associated with the calculation of the limits of the classes. Indeed, this stage may sometimes be rendered difficult when there is limited knowledge. Expert opinions have to be sought, with the subjectivity that this implies. The

other limitation of this method is linked to potential existence of a "threshold effect". For values close to the threshold, a small variation in class change could lead to a large variation in the final result. Conversely, a large variation that is confined to the same class will have no effect on this result.

This explains most of major difficulties that confronted us in this method with regard to the validation of mapping results. The aggregation process could be improved with the precise qualification of each category of risk. Including multi-criteria evaluation processes in a GIS software may consequently be useful, leading to some potentially interesting future research topics of these researches.

5.4.5. A method restricted to a specific natural context

Our method is restricted to areas of low permeability substratum where water and pollutant fluxes are mainly exported by surface and subsurface flows. It cannot be used in its current form in karstic areas, for example, where infiltration is more predominant. It must therefore be adapted by incorporating vulnerability variables for the environment itself into the pollution of groundwater and not just surface water. It would no longer be merely a question of considering the distance to water courses, but to the groundwater as well. The difficulty would reside in evaluating the proportions of transfer into surface water and groundwater. The variables for geological substratum and soils would therefore become essential.

6. Conclusions and outlooks

This research has focused on the development of a method for estimating agricultural pesticide contamination risks in surface waters, and the aggregation of risk data to correspond to various spatial organisational levels. The method provides a decision-making tool for public managers in charge of water quality for human consumption and environmental sectors. Different spatial scales were investigated, conserving the initial data resolution throughout the entire process. Classification thresholds for the levels of risk were used for the relative ranking of risk areas. The final maps produced in our study area are based on precise data collected in the field and are designed to correspond to appropriate scale for public environmental managers (allowing for the definition of priority action zones). Agri-environmental measures required to re-establish the water quality in the environment, can then be targeted.

The validation of the method was performed by comparison with field biological data, a measurement of pollution effects on the aquatic environment. We compared the contamination risks calculated with biological (diatom) responses, determined from simultaneously performed surveys. Biological data were correlated with the risk levels determined by the PHYTOPIXAL method, providing an indirect, but significant, validation of the method.

The large-scale, high spatial resolution of the method allows for the coverage of pollution risks at different scales and therefore constitutes an interesting alternative to traditional diagnostic studies, which are time-consuming and costly.

Future developments would be to improve the aggregation process of criteria, conceivably using a multi criteria evaluation function integrated in GIS, to be able to assess contamination risks over a large area. The links between the results of the risk zoning for non-point pollution of agricultural origin and aquatic organisms could be confirmed, either using a battery of test organisms (e.g. combining diatoms and benthic macro-invertebrates) that are integrative of various anthropogenic impacts at different time scales, or by confirming the present results in other watersheds.

However, this work can be already of practical interest to the environmental managers in different countries, to establish zoning priorities for the implementation of precisely targeted agri-environmental water protection measures.

References

Alkan Olsson, J., Bockstaller, C., Stapleton, L.M., Ewert, F., Knapen, R., Therond, O., Geniaux, G., Bellon, S., Correira, T.P., Turpin, N., Bezlepkina, I., 2009. A goal oriented indicator framework to support integrated assessment of new policies for agri-environmental systems. Environ. Sci. Policy 12, 562–572.

Arnold, J., Srinivasan, R., Muttiah, R.S., Williams, J.R., 1998. Large-area hydrologic modelling and assessment. J. Am. Water Resour. 34, 73–89.

Aspinall, R., Pearson, D., 2000. Integrated geographical assessment of environmental condition in water catchments: Linking landscape ecology, environmental modelling and GIS. J. Environ. Manage. 59, 299–319.

Aurousseau, P., Gascuel-Odoux, C., Squividant, H., 1998. Éléments pour une méthode d'évaluation d'un risque parcellaire pour la contamination des eaux superficielles par les pesticides. Application au cas de la contamination par les herbicides utilisés sur culture de maïs sur des bassins versants armoricains. Étude et gestion des sols 5 (3), 14.

Balram, S., Dragicevic, S., Feick, R., 2009. Collaborative GIS for spatial decision support and visualization. J. Environ. Manage. 90, 1963–1965.

Barriuso, E.coord, 2004. Estimation des risques environnementaux des pesticides. INRA Ed.

Beaujouan, V., Durand, P., Ruiz, L., Aurousseau, P., Cotteret, G., 2002. A hydrological model dedicated to topography-based simulation of nitrogen transfer and transformation: rationale and application to the geomorphology–denitrification relationship. Hydrol. Process. 16, 493–507.

Bechmann, M.T., Krogstad, T., Sharpley, A.N., 2005. A phosphorus index for Norway. Acta Agr. Scand. 55 (3), 205–213.

Blöschl, G., Sivapalan, M., 1995. Scale Issues in Hydrological Modelling: A Review. John Wiley & sons, Chichester, GBR.

Bock, M., Rossner, G., Wissen, M., Remm, K., Langanke, T., Lang, S., Klug, H., Blaschke, T., Vrscaj, B., 2005. Spatial indicators for nature conservation from European to local scale. Ecol. Indic. 5, 322–338.

Bockstaller, C., Girardin, P., Van der Werf, H.M.G., 1997. Use of agro-ecological indicators for the evaluation of farming systems. Eur. J. Agric. 7, 261–270.

Boithias, L., Sauvage, S., Taghavi, L., Merlina, G., Probst, J.L., Sanchez-Perez, J.M., 2011. Occurrence of metolachlor and trifluralin in the Save river agricultural catchment during floods. J. Harzard. Mater. 196, 210–219.

Brisson, N., Gary, C., Justes, E., Roche, R., Mary, B., Ripoche, D., et al., 2003. An overview of the crop model STICS. Eur. J. Agric. 18 (3–4), 309–332.

Buczko, U., Kuchenbuch, R.O., 2010. Environmental Indicators to assess the risk of diffuse nitrogen losses from agriculture. Environ. Manage. 45, 1201–1222.

Carluer, N., Tournebize, J., Gouy, V., Margoum, C., Vincent, B., Gril, J.J., 2009. Role of buffer zones in controlling pesticides fluxes to surface waters. Proc. Environ. Sci. 9, 21–26.

Cemagref, 1982. Étude des méthodes biologiques d'appréciation quantitative de la qualité des eaux. Rapport Q.E. Lyon A.F. – Bassin Rhône-Méditerranée-Corse.

Cemagref, 2006. Digital soil map of the Save obtained from the map prepared by pedologists of the CACG in the 1960s. In: Cemagref, UR ADBX, Bordeaux, p. 1.

CEE, 1991a. Directive n°91/414/CEE du 15/07/91 concernant la mise sur le marché des produits phytopharmaceutiques. JOCE n°L 230 du 19 août 1991 et rectifié au JOCE n°L 170 du 25 juin 1992.

CEE, 1991b. Directive du conseil n°91/676/CEE du 12 décembre 1991 concernant la protection des eaux par les nitrates à partir de sources agricoles. JOCE n°L 375 du 31 décembre 1991.

CEE, 1998. Directive n°98/83/CE du 03/11/98 relative à la qualité des eaux destinées à la consommation humaine. JOCE n°L 330 du 5 décembre 1998 et rectifié au JOCE n°L 111 du 20 avril 2001.

CORPEN (Ed.), 2003. Diagnostic régional de la contamination des eaux liée à l'usage des produits phytosanitaires: éléments méthodologiques. Ministères en charge de l'agriculture et de l'environnement, Paris, p. 84.

Cotter, A.S., Chaubey, I., Costello, T.A., Soerens, T.S., Nelson, M.A., 2003. Water quality model output uncertainty as affected by spatial resolution of input data. J. Am. Water Resour. Assoc. 39 (4), 977–986.

Dalgaard, T., Hutchings, N.J., Porter, J.R., 2003. Agroecology, scaling and interdisciplinarity. Agri. Ecosyst. Environ. 100, 39–51.

Deneux, M., Emorine, J.P., 1998. Avenir de la réforme de la Politique Agricole Commune. Report No. 466. Sénat, Paris.

Devillers, J., Farret, R., et al., 2004. Indicateurs pour évaluer les risques liés à l'utilisation des pesticides. ouvrage collectif. Lavoisier, Editions Tec et Doc.

Dixon, B., Earls, J., 2009. Resample or not? Effects of resolution of DEMs in watershed modeling. Hydro. Process. 23 (12), 1714–1724.

Ducrot, D., 2005. Méthodes d'Analyses et D'interprétation D'images Multi-Sources. Extraction de Caractéristiques du Paysage, Mémoire de Recherche. CESBIO, Toulouse.

Dumanski, J., Pettapiece, W., McGregor, R.J., 1998. Relevance of scale dependent approaches for integrating biophysical and socio-economic information and development of agroecological indicators. Nutr. Cycl. Agroecosyst. 50, 13–22.

Eastman, J.R., 2001. Idrisi32 Release 2 Tutorial. Clarklabs, Worcester.

Eastman, J.R., Kyem, P.A.K., Toedano, J., Jin, W., 1993. Explorations in Geographic Systems Technology, 4, GIS and Decision Making. UNITAR, Geneva.

Ernoult, A., Bureau, F., Poudevigne, I., 2003. Patterns of organisation in changing landscapes: implications for the management of biodiversity. Landscape Ecol. 18, 239–251.

EC, 2000. Directive 2000/60/EC of the European Parliament and of the Council of 23 October 2000 Establishing a Framework for Community Action in the Field of Water Policy.

FAO, 2006. World Reference Base for Soil Resources, A Framework for International Classification, Correlation and Communication, World Soils Resources Report 106.

Ferrant, S., Oehler, F., Durand, P., Ruiz, L., Salmon-Monviola, J., Justes, E., Dugast, P., Probst, A., Probst, J.L., Sanchez-Perez, J.M., 2011. Understanding nitrogen transfer dynamics in a small agricultural catchment: Comparison of a distributed (TNT2) and a semi distributed (SWAT) modeling approaches. J. Hydrol. 406, 1–15.

Gardi, C., 2001. Land use, agronomic management and water quality in a small Northern Italian watershed. Agric. Ecosyst. Environ. 87, 1–12.

Gascuel-Odoux, C., Aurousseau, P., Cordier, M.O., Durand, P., Garcia, F., Masson, V., Salmon-Monviola, J., Tortrat, F., Trepos, R., 2009a. A decision-oriented model to evaluate the effect of land use and agricultural management on herbicide contamination in stream water. Environ. Modell. Softw. 24, 1433–1446.

Gascuel-Odoux, C., Massa, F., Durand, P., Merot, P., Troccaz, O., Baudry, J., Thenail, C., 2009b. Framework and tools for agricultural landscape assessment relating to water quality protection. Environ. Manage. 43, 921–935.

Gouy, V., Gril, J.J., 2001. Diagnosis of pesticide diffuse pollution and management practices to reduce transfer to water. Ing. EAT, special pesticides: 81–90.

Henin, S., 1980. Rapport du groupe de travail activités agricoles et qualité des eaux. Synthèse. Ministères en charge de l'agriculture et de l'environnement, pp. 59.

Jiang, H., Eastman, J.R., 2000. Application of fuzzy measures in multi-criteria evaluation in GIS. Int. J. Geogr. Inf. Sci. 14 (2), 173–184.

Le Bissonnais, Y., Montier, C., Daroussin, J., King, D., 1998. Cartographie de l'aléa "Erosion des sols" en France. Collection Etudes et Travaux n° 18. IFEN, INRA, Orléans.

Lepinard, P., 2008. Les méthodes de discrétisation dans ArcGIS 9.2. Didacticiel de formation ESRI, 45 pp.

Lescot, J.M., Bordenave, P., Petit, K., Leccia, O., Sanchez-Perez, J., Sauvage, S., Probst, J.L.,2011. Cost-effectiveness analysis for controlling water pollution by pesticides using SWAT and bio-economical modelling. In: SWAT International Conference. University of Castilla la Mancha, Toledo, Spain.

Lim, K.J., Engel, B.A., 2003. Extension and enhancement of national agricultural pesticide risk analysis (NAPRA) WWW decision support system to include nutrients. Comput. Elect. Agric. 38, 227–236.

Macary, F., Vernier, F., 2007a. Zonage de risque potentiel de transferts de pesticides à l'échelle de bassins versants: quelles méthodes pour un transfert d'échelles spatiales? Pesticides: impacts environnementaux, gestion et traitements. Presses de l'Ecole Nationale des Ponts et Chaussées, pp. 245–259.

Macary, F., Balestrat, M., Puech, C., Saudubray, F., 2007. In: Batton-Hubert, M., Joliveau, T., Lardon, S. (Eds.), Méthode d'estimation d'un risque potentiel spatialisé de la contamination des eaux de surface par les intrants agricoles à différentes échelles spatiales: application sur les coteaux de Gascogne. SAGEO, Rencontres Internationales Géomatique et Territoire.

Macary, F., Leccia, O.coord, et al., 2011. European Project AGUAFLASH: Deliverable GT3.2. Identification of Contamination Risk Areas of Surface Waters on the Watersheds of Flumen (Spain), Alegria (Spain) and Enxoé (Portugal).

Malczewski, J., 1999. GIS and Multicriteria Decision Analysis. John Wiley and Sons, New York.

Malczewski, J., 2006. GIS-based Multicriteria decision analysis: a survey of the literature. Int. J. Geogr. Inf. Sci. 20 (7), 703–726.

Maystre, L.Y., Pictet, J., Simos, J., 1994. Méthodes Multicritères ELECTRE: Description, Conseils Pratiques et Cas D'application à la Gestion Environnementale. Lavoisier, TEC and DOC, Paris.

Mérot, P., Gascuel-Odoux, C., Walter, C., Zhang, X., Molénat, J., 1999. The influence of hedgerow networks in bocage landscapes on surface water pathways. J. Water Sci. 12, 23–44.

Misra, A.K., Baker, J.L., Mickelson, S.K., Shang, H., 1996. Contributing area and concentration effects on herbicide removal by vegetative buffer strips. Am. Soc. Agric. Biol. Eng. 39, 2105–2111.

Morin, S., Bottin, M., Mazzella, N., Macary, F., Delmas, F., Winterton, P., Coste, M., 2009. Linking diatom community structure to pesticide input as evaluated through a spatial contamination potential (Phytopixal): a case study in the Neste river system (South-West France). Aquat. Toxicol. 94, 28–39.

Paegelow, M., Olmedo-Camacho, M.T., 2005. Possibilities and limits of prospective GIS land cover modelling- a compared case study: Garrotxes (France) and Alta Alpujarra Granadina (Spain). Int. J. Geogr. Inf. Sci. 19 (6), 697–722.

Paegelow, M., Camacho-Olmedo, M.T., 2008. Modelling Environmental Dynamics: Advances in Geomatics Solutions. Springer, Berlin.

Patty, L., Réal, B., Gril, J.J., 1997. The use of grassed buffer Strips to remove pesticides, nitrate and soluble phosphorus compounds from runoff water. Pest. Sci. 49 (3), 243–251.

Perrin, A.S., Probst, A., Probst, J.L., 2008. Impact of nitrogen fertilizers on carbonate dissolution in small agricultural catchments: implications for weathering CO_2 uptake at regional and global scales. Geochim. Cosmochim. Acta 72, 3105–3123.

Probst, J.L., Tardy, Y., 1985. Fluctuations hydroclimatiques du Basin d'Aquitaine au cours des 70 dernières années. Rev. Geol. Dyn. Geogr. Phys. 26 (1), 59–76.

Puech, C., Dartus, D., Bailly, J.S., Estupina-Borrell, V., 2003. Hydrologie distribuée, télédétection et problèmes d'échelle. Société française de photographie et télédétection 172, 11–21.

Revel, J.C., Guiresse, M., 1995. Erosion due to cultivation of calcareous clay soils on the hillsides of south west France. Effect of former farming practices. Soil Till. Res. 35 (3), 147–155.

Riley, J., 2001. Indicator quality for assessment of impact of multidisciplinary systems. Agric. Ecosyst. Environ. 87, 121–128.

Riitters, K.H., 2005. Downscaling indicators of forest habitat structure from national assessments. Ecol. Indic. 5, 273–279.

Roy, B., 1990. The outranking approach and the foundations of ELECTRE methods. In: Bana, C.A. (Ed.), Readings in Multiple Criteria Decision Aid. Springer-Verlag, Heidelberg.

Saaty, T.L., 1990. How to make a decision: the analytic hierarchy process. Eur. J. Operat. Res. 48, 9–26.

Schärlig, A., 1996. Pratiquer Electre et Prométhée. Un complément à Décider sur plusieurs critères. Presses polytechniques et universitaires romandes, Lausanne, pp. 176.

Schmitt, T.J., Dosskey, M.G., Hoagland, K.D., 1999. Filter strip performance and processes for different vegetation, widths, and contaminants. J. Environ. Qual. 28, 1479–1489.

Schröder, J.J., Scholefield, D., Cabral, F., Hofman, G., 2004. The effects of nutrient losses from agriculture on ground and superficial water quality: the position of science in developing indicators for regulation. Environ. Sci. Policy 7, 15–23.

Seguin, F., Druart, J.C., Le Cohu, R., 2001. Effects of atrazine and nicosulfuron on periphytic diatom communities in freshwater outdoor lentic mesocosms. Ann. Limnol. 37, 3–8.

Shannon, C.E., Weaver, W., 1949. The Mathematical Theory of Communication. Univ. Illinois Press, Urbana, USA.

Simos, J., 1990. Evaluer l'impact sur l'environnement: Une approche originale par l'analyse multicritère et la négociation. Presses polytechniques et universitaires Romandes, Lausanne.

Stein, A., Riley, J., Halberg, N., 2001. Issues of scale for environmental indicators. Agric. Ecosyst. Environ. 87, 215–232.

Srinivasan, R., Arnold, J.G., 1994. Integration of a basin-scale water quality model with GIS. Water Res. 30, 453–462.

Surgan, M., Condom, M., Cox, C., 2010. Pesticide risk indicators: unidentified inert ingredients compromise their integrity and utility. Environ. Manage. 45, 834–841.

Taghavi, L., Merlina, G., Probst, J.L., 2011. The role of storm flows in concentration of pesticides associated with particulate and dissolved fractions as a threat to aquatic ecosystems. Knowl. Manag. Ec. 400 (06), 1–11.

Tong, S.T.Y., Chen, W., 2002. Modeling the relationship between land use and superficial water quality. J. Environ. Manage. 66, 377–393.

Van Cauwenbergh, N., Biala, K., Bielders, C., Brouckaert, V., Franchois, L., Garcia Cidad, V., Hermy, M., Mathijs, E., Muys, B., Reijnders, J., Sauvenier, X., Valckx, J., Vanclooster, M., Van der Veken, B., Wauters, E., Peeters, A., 2007. SAFE-A hierarchical framework for assessing the sustainability of agricultural systems. Agric. Water. Manage. 120, 229–242.

Van der Werf, H.M.G., 1996. Assessing the impact of pesticides on the environment. Agric. Water. Manage. 60, 81–96.

Vazquez-Amabile, G., Engel, B.A., 2008. AGIRISK-swat: A decision support system to compute hydrologic and nps pollution risk analysis in agricultural watersheds. American Society of Agricultural and Biological Engineers. In: Conference on 21st Century Watershed Technology: Improving Water Quality and Environment, Concepción, pp. 462–470.

Xinhao, W., Yin, Z.Y., 1997. Using GIS to assess the relationship between land use and water quality at a watershed level. Environ. Int. 23, 103–114.

Wesseling, J.G., Feddes, R.A., 2006. Assessing crop water productivity from field to regional scale. Agric. Water. Manage. 86 (1–2), 30–39.

Weyerbrock, S., 1998. Reform of the European Union's Common Agricultural Policy: how to reach GATT-compatibility? Eur. Econ. Rev. 42, 375–411.

Williams, J.R., 1990. The erosion-productivity impact calculator (EPIC) model: a case history. Philos. Trans.: Biol. Sci. 329, 421–428.

Wohlfahrt, J., Colin, F., Assaghir, Z., Bockstaller, C., 2010. Assessing the impact of the spatial arrangement of agricultural practices on pesticide runoff in small catchments: Combining hydrological modeling and supervised learning. Ecol. Indic. 10 (4), 826–839.

Wood, E., Sivapalan, M., Beven, K., Band, L., 1988. Effects of spatial variability and scale with implications to hydrologic modeling. J. Hydrol. 102, 29–47.