Challenges for integrated assessment and cost-effectiveness analysis of mitigation measures for controlling water pollution

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ABSTRACT

Identification of challenges for integrated assessment of mitigation measures proposed to improve the ecological and chemical status of water bodies is explored in this paper. These challenges are addressed within a framework proposed for a spatially-distributed cost and effectiveness analysis. Resulting environmental and economic impacts of the implementation of measures are assessed using agro-hydrological and bio-economic modelling with effectiveness and costs calculated at each sub basin level. Cost-effectiveness ratios calculated over a long period of measures implementation depend on the location where measures are applied. Mapping costs and effectiveness at the basin sub-level reveals the locations where implementing a measure (within interval confidence) would be the most cost-effective. To compare measures between themselves, cost-effectiveness ratios can also be calculated for the whole catchment by adding up sub basin total costs, and by assessing effectiveness at the watershed outlet. Advantages and shortcoming of the integrated approach are discussed with challenges to investigate all the uncertainties related to mitigation programmes of agro environmental policies for reducing water pollution.

KEY WORDS: *Integrated assessment, Cost Effectiveness analysis, Mitigation measures, Watersheds*.

INTRODUCTION

When formulating programmes of measures to be implemented in the European river basins, the EU water Framework Directive proposed the Cost-Effectiveness Analysis (CEA) as an economic tool for minimizing costs, effectiveness being supposed to be implied. The approach to CEA is however not specified and no standardized approach to CEA is proposed. The aim of this paper is to discuss and evaluate a pragmatic quantitative approach based on a spatially distributed Cost-Effectiveness Analysis (sCEA) applied on two case studies within the Garonne river basin (Upstream Gers river basin) and the Charente river basin (Ne watershed) in terms of their applicability, transparency and capability for decision-making. Conclusions are drawn with regard to the suitability of the approaches and the uncertainties.

Our approach detailed in Lescot, 2013, is integrated because of its interdisciplinary nature implying coherence and consistency of the linkages between models. It could be categorized as a bottom-up approach focusing on the cost of implementing measures at the micro level (hydrological unit) and the macro level (watershed) while assessing environmental effectiveness at the sub and watershed level (Figure 1). The effectiveness of some remediation-mitigation measures can be assessed with regard to intermediate goals (reduction of pressure) by the use of indicators and/or final goals (impacts) where models are needed. Given the objectives laid down in the WFD (water bodies to be returned to “good” ecological status by 2015), impacts need to be evaluated. In addition, pressure indicators[1] for pesticides although widely used in policy analysis could back up model outcomes only if they are spatially distributed.

Questions of spatial, temporal, and technological heterogeneity make devising environmental mitigation programmes a complicated exercise since the characteristics of agricultural production such as soil types, slopes, farming systems or proximity to streams can vary hugely across a river basin. Furthermore, the precise extent of the damage caused to the environment by the use of pesticides is difficult to assess, due to the delay between their application and the appearance of any quantifiable effects. In addition, these relationships are subject to a number of stochastic influences outside the farmers’ control.

Finding the right scale has been identified as one of the major challenges, conceptually and methodologically in all science that uses geographic information. For assessing mitigation measures, the scale to select is neither straightforward nor neutral. Nonetheless, to carry out a CE analysis, costs and effectiveness should best be calculated at a common scale appropriate to represent underlying physical processes. Because the watershed or sub basin level is the suitable scale for assessing effectiveness in term of pollutant concentration reduction in outlets, the costs should be assessed at the same scale which may cause difficulties because of the lack of information on farms plots location and practices. In addition to space scale, there should be coherence and consistency for time scale and the choice of the appropriate time horizon. Hydrological modelling offers a means of assessing impacts over a long period of time covering several years of implementation of measures. Beyond that, it is generally advocated that results obtained by modelling are an objective source of information that can be used to support decisions.

To assess the impacts of mitigation strategies, we use the SWAT model (Neitsch *et al.*, 2005) with ArcSWAT[2]. The SWAT model is suitable for predicting long-term impacts of mitigation measures on agricultural chemical yield and for simulating agricultural management practices. The overarching objective of this paper is to address challenges

[1] Such as EIQ (environmental Impact Quotient) or QSA (Quantity of active substances sold), NODU (Number of Used Doses) and IFT (Treatment frequency index) used in France.

[2] Geographic Information System ArcGIS10 interfaced with SWAT 2009.

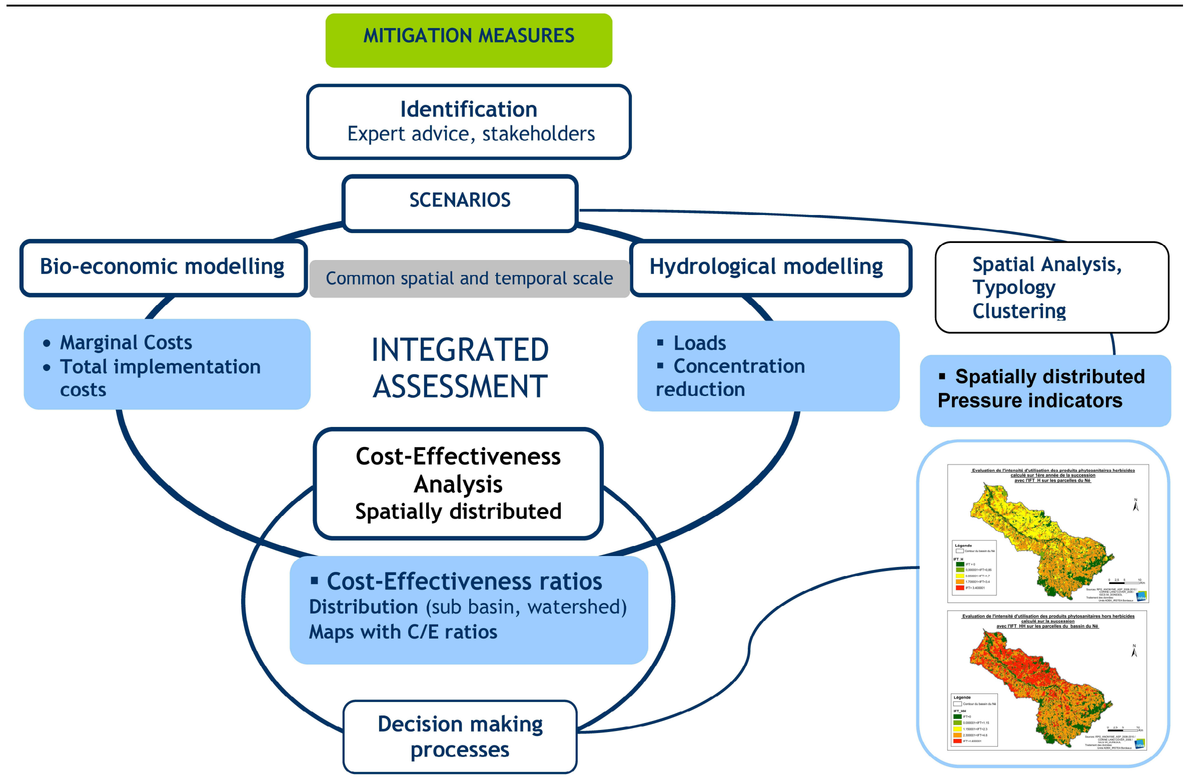


Figure 1. Integrated assessment developed for spatially distributed cost-effectiveness analysis.

in integrated assessment of mitigation measures related to the following elements (Figure 1):

- The economic modelling to assess marginal costs of implementing mitigation measures at the hydrologic unit (HRU)[3] and quantify total costs at the catchment level over a period similar with the hydrological simulation period,
- The hydrological modelling to assess over a long period the pollution reduction of pollutants in water streams following implementation of measures,
- The defining of scenarios: Spatial Analysis, Clustering, typologies to assess standard crop rotations and practices and to help build spatially distributed pressure or combined pressure/vulnerability indicators,
- Interaction with stakeholders, communities and governance systems.

CHALLENGES

Limitations of approaches for costs calculation

When choosing mitigation measures, their effectiveness is supposed to be implied. However, if we assume that two measures could have the same effectiveness in reducing pollution, this can be theoretically obtained at different cost because marginal costs for implementing measures are not equal between themselves, given a location, and for a same measure between locations where it is implemented). Farm models allows traditionally for modelling behaviour of individual farms but requires wide information that is often not available. Using representative farms or type farms is an attempt to overcome these difficulties. Nonetheless, by doing so, we loose geographic information on farms plots and localization of practices. In addition, technological heterogeneity is not addressed with representative (average) farms and not completely with type (modal) farms.

In many studies, the loss in total gross margin is calculated either by a linear programming (LP) model either by partial budgeting. LP models usually (e.g. Volk, 2008), optimise agricultural production programmes under different management scenarios and costs are assessed by comparison of gross margins. On the other side, econometric approaches can help assess the estimated changes in the gross profit margins of a large data set of farms (e.g., Fezzi, 2001). Moreover, econometric models automatically provide estimates of uncertainty derived directly from statistical inference rather than requiring sensitivity analysis. Nonetheless we preferred the methodology of economic optimisation, more relevant when dealing with limited and incomplete information and able to link economic elements with ecological and biophysical elements.

The alternative form of aggregation we used, which overcomes above scale problems, combines farm plots in

[3] Hydrological Response Units (HRU), consisting of homogeneous land use, management and soil characteristics, and therefore different in size. HRUs are the spatial units used by the SWAT model to calculate the vertical, lateral and sub-lateral flows of water and nutrients .These flows are then aggregated for each sub basin. Water and pesticides from HRUs in sub watersheds are routed to the sub watershed outlets.

each spatial unit together, treating them as a single entity ensuring consistency with the environmental modelling. Each HRU is then treated as a single entity with a crops sequence. Yields of crops provided by the SWAT model are used in the bio economic model so that to take into account their heterogeneity related to diverse practices and soil types.

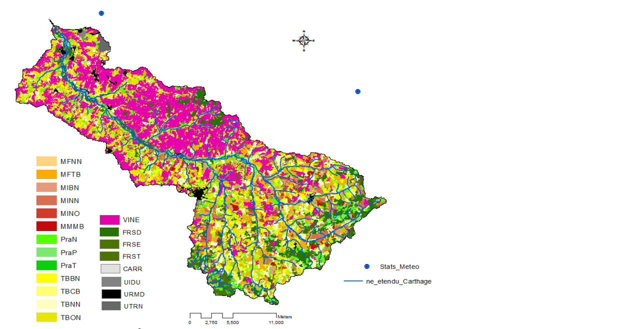
Through the linear input/output analysis, the Leontief production functions (vectors of fixed coefficients) provide the possibility for a description of the production possibility set within a HRU. For each output, there are a number of techniques (crops and practices with and without measures) in competition, subject to constraints from the availability of inputs and/or attached to measures. As a result of optimisation, each combination of constraint levels and production techniques results in a single solution. Financial incentives are needed to compensate extra costs and encourage adoption of measures. When the decision variable representing activity with measure appears in optimum solution, incentive and marginal (shadow) costs cancel each other out. Optimisations runs allow producing implementation cost curves by HRU. Total cost of implementation is calculated by integration of shadows costs over the area where the measure can be implemented[4]. Costs calculated by HRU and on a one-year basis are then added at the sub basin level. For the period of measures implementation, we use the discounted sum of annual costs defined as follows:

$$\left(\sum_{t=1}^{T} AC\left(1+i\right)^{-t}\right) \qquad (1)$$

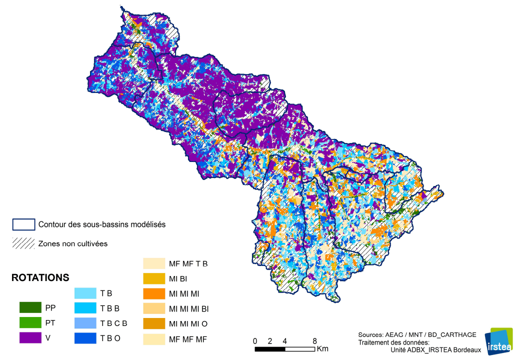
AC: annual cost (€); T: years of the simulation period; i: discount rate.

Such costs could be calculated with an inter-temporal linear programming model but this implies taking into account uncertainty on prices over a long period. Because decision making for adoption (or not) of measures tends to be determined on a short-term horizon, the one-year basis calculation is more appropriate. This choice is nevertheless questionable when measures are related to structural changes e. g. changing arable land to grassland or implementing buffer strips. Developed with the General Algebraic Modelling System (GAMS) (Brooke *et al*., 1988; Mc Carl, 2009) the economic model simulates agricultural land use at each HRU level. As it is assumed that farmers are price-takers and profit maximisers, the economic model maximises expected utility by choosing whether or not to implement a particular measure. Agro-environmental measures are introduced into the bio-economic farm model either as new activities or by modifying the parameters for practices. Initially, we worked on typical crops rotation defined by Principal Component Analysis and Cluster Analysis on Land use and management sequences to identify (administrative) spatial unit with homogeneous characteristics with respect to the crops and practices. The main crop rotation identified by spatial unit is considered to be implied on its entire area.

[4] In addition, model outcomes reveal the changes in land use when measures are gradually implemented with increasing incentives. This information is in a way a side-product of the economic model as changes in practices may impact other production choices to maximise profit. Simulations could be used to study the trade-offs between implemented measures and land use changes at the HRU, sub-basin and basin levels for increasing levels of incentive



Land use



Rotations defined from RPG and used by GENLU2

Figure 2. Shift from land use to rotations (Ne case study)

In its last development and in order to be consistent with the SWAT model, the economic model uses henceforth rotations and practices as defined by a model from Irtsea named GENLU2. GENLU2 develops and generates land use dynamics by shifting from land use to rotations.

This model uses decision rules and expertise to randomly create crops sequences and practices (with and without measures) spatially distributed within the watershed at the “RPG block” scale[5].

GENLU2 uses as an entry the files provided by a previous work performed on the area to describe in a simplified way the agricultural systems and the practices. A typology of soils, agricultural systems and practices is made using clustering techniques (Vernier *et al.*, 2010). The main rotations are defined for each type of soil with the percentage of the area dedicated to each rotation. Then average practices are described for each crop inside the rotation. The RPG data are used to spatially distribute the typology. A combination of soil type-crop rotation-crop is allocated to each RPG block. The result of this work is twofold: first a detailed map of the rotations on the area (Figure 2) with a table of the percentage of each rotation by type of soil used by GENLU2 as decision rules for the generation of the SWAT Land use files (Figure2) and second, a table with the detailed practices for each crop in

[5] Registre Parcellaire Graphique (RPG): Parcel referencing system used in the French Land Parcel Identification System in the frame of the European Council Regulation No 1593/2000 requiring. A RPG (or CAP) block is the elementary spatial unit that groups together a number of neighbouring agricultural parcels with one or more crops cultivated by one farmer and delineated by the most stable boundaries.

the rotation used by GENLU2 to generate the SWAT management files (mgt). These tables are also used as an entry for the calculation of spatial environmental indicators (a part of the integrated assessment process not detailed in this paper). Percentages of area for each crop and each type of soils within the watershed limits are validated afterwards by expertise. Files generated by this model are able to be directly used by the SWAT model. Because costs are calculated on a one-year basis with the bio-economic model, the shifting from rotations to land use is carried out by allocating land of the HRU to crops proportionally to their part in the crops sequence over the rotation period.

Limitations of approaches for hydrological modelling

Once the SWAT model is updated and calibrated, it can be used to simulate the pesticide mitigation strategies and evaluate their effectiveness. The effectiveness is presented in terms of absolute reduction in pesticide concentrations (in $\mu g.l^{-1}$) following the implementation of a measure. The first step of the model is to subdivide the watershed into sub basins that are further disaggregated into Hydrological Response Units (HRU). The management practices are defined at the HRU level by specific management operations (beginning and end of growing season, timing of cultural operations, amount of fertilizers and pesticides, irrigation management). Loads then concentrations of nutrients and pesticides are calculated at the outlet of the whole watershed, and of each sub basin from the SWAT main channel output files with reference concentration of the baseline scenario. To consider the problem of pesticide time lags (Roa-García & Weiler, 2010), simulations are performed over a 25-year period with measures being applied each year. For the final pesticide concentration, we retained the average value of pesticide concentrations calculated over the last ten years of the modelling period.

Concentration reduction is calculated as the difference between these average concentrations:

$$\left([C_0]-[C_s]\right) \tag{2}$$

$[C_s]$: average concentration over the ten last years of hydrological simulation ($\mu g.l^{-1}$) with measure applied each year (for a given area); $[C_0]$: baseline concentration ($\mu g.l^{-1}$).

For water flows, predicted values generally matched well with the observed values for both calibration and validation periods (coefficient of determination and Nash-Suttcliffe Efficiency index). Uncertainties on the irrigation water volumes actually applied by farmers and the lack of detailed information on the management of low water regimes by water agencies may sometimes explain discrepancies between simulated and observed daily values.

Modelling pesticide run-offs is more complicated because of the number of active substances and diverse protection strategies. In our case studies, we restricted the number of active substances to the most widely used molecules and/or the molecules the most frequently found in sampling water analysis. The set of active substances[6] applied to each main crop of the watershed is listed with the frequency of their use either from expertise (Ne watershed) either from surveys carried out on farms with additional expertise (Upstream Gers river basin). In this late case, we defined new “average active ingredients” (AAI) based on physical and chemical properties[7] (Koc, DT50 and solubility) of the molecules for each pesticides group with the average dose (arithmetic mean) applied by farmers. In the Ne watershed study, each of the ten widely used molecules has been simulated separately. Average simulated concentrations and measures occasionally differ significantly and these differences between simulated and measured concentrations are difficult to handle because of the uncertainty of the measurements themselves (related to their low frequency and the limited number of values within a month and year).

Limitation in the use of Cost-Effectiveness ratios

CEA is an appropriate approach for evaluating mitigation measures where the measurement of benefits is difficult and unsure. CEA cannot determine the overall value of a single measure but is particularly useful in comparing two or more measures. It summarizes results into single useful quantitative indicators for selecting measures. For the sCEA, we used a spatialized cost-effectiveness indicator R, defined at the sub basin or watershed level, given the area implemented with the measure. Thus, for a given area implemented with measures we have:

$$R_{subbasin/basin} = \frac{discounted\ sum\ of\ annual\ costs\ (€)}{concentration\ reduction\ (\mu g.l^{-1})} \tag{3}$$

This ratio enables measures to be ranked in terms of increasing unit costs per unit of pesticide concentration reduction, given the range of uncertainty. Calculated at the sub basin level and represented on maps, they allow for rational discussion between stakeholders who often require integrated information. Such maps give an overview of the cost and effectiveness of the various possible mitigation measures, depending on the location where they could be implemented and to which extend WFD objectives could be reached.

MANAGING UNCERTAINTY

A lack of scientific knowledge and empirical evidence are part of the difficulties in the political negotiation process surrounding the formulation of concrete water policy objectives, including indicator parameters and specific threshold values[8] (Brouwer & Blois, 2008). Uncertainties can be found at each step of our analysis. There is first uncertainty relating primarily to the identification of the environmental objectives (restoration of good ecological status) and their monitoring (EU health-based drinking water

[6] All pesticides that are officially registered for use and are likely to be used in the watershed area should be monitored and the total sum calculated by adding all concentrations that exceed the parametric value/detection limit of 0.1 µg.l-1. Thus there is no standard about the number of pesticides that should be considered in total as this will vary between watershed areas and could add up to hundreds of pesticides.

[7] Koc: Sorption coefficient normalized to organic carbon content; DT50:Time for a 50% decline of the initial pesticide concentration

[8] The value of $0.1\mu g.l^{-1}$ in the WFD is a substitute for zero i.e. absence in water or below the detection limit. The parametric values for individual pesticides ($0.1\mu g.l^{-1}$) and for total pesticides ($0.5\mu g.l^{-1}$) are not based on any scientific findings. Besides, the World Health Organization uses a different approach with different set of guideline values for a large number of individual pesticides.

pollutant concentration limits and method detection limits). There is next uncertainty relating to the identification of the main pollution sources (point and non-point) and of their contribution to the water quality problem (agriculture, gardening, ditches or railways maintenance).

Point-source pollution by pesticides mainly relates to pesticides handling (filling, cleaning, remnant management) while the non-point pollution sources are more related to the application of pesticides and often result from natural environmental factors (runoff, drift, etc.). Few studies are available on the subject, in which point and non-point sources are clearly separated. In the literature, it is estimated that point sources could contribute up to 50% to the pollution of surface water by pesticides (Leu *et al*., 2004; Bach *et al*., 2005). Moreover average practices do not take account of excessive use of pesticides.

There is uncertainty also relating to the identification of potential mitigation measures proposed by science to solve the water-quality problem. Effectiveness is firstly tested on the plot scale and only assessed by modelling on a large scale). Moreover, best location for implementing the measures is not straightforward as the zones identified by the pollution pressure may not contribute the most to pollution at the outlet. Finally there are uncertainties in the modelling itself due to the availability of data (limited water analysis and not always at appropriate times of flow for calibration and validation processes) and uncertainty on values. Sensitivity analysis is usually carried out when using the SWAT model for selected parameters and different management practices. Classification of existing methods of sensitivity analysis refers to the way parameters are treated: local techniques concentrate on estimating the local impact of a parameter on the model output while global techniques analyse the whole parameter space at once (van Griensven *et al*., 2006). These authors propose a sampling strategy allowing a global sensitivity analysis with only limited number of model runs overcoming the problem of over parameterization for distributed model like the SWAT model.

Uncertainties and error margins in the estimation of the costs should be considered as well with bio economic modelling. In sensitivity analysis, consideration is given to the way in which errors in a set of input data affect the error in the final output. In practice we could suppose that varying the number of constraints will have no effect on the costs calculation. On the contrary, varying the number of activities by proposing other new crops with environmentally friendly practices may change the costs calculated. Much attention should be paid also to the change of the objective function (minimizing risk instead of maximizing profit), the values for sub optimal strategies or the variation of uncertain technical parameters (such as prices of inputs and outputs) that would help to observe changes in calculated costs of mitigation measures. Because the bio economic model is used for comparison of costs and not for their absolute values, we could suppose that errors in input data affect the costs calculation the same way for the different measures.

Questions arise lastly with the use of the CE indicator on its level of confidence as sensitivity analysis both for hydrologic and economic/economical assessments will lead to a range of values determined by the way how parameters will be treated.

CONCLUSION

J. D. Brown, 2004 argues in favour of explicit assessments of uncertainty in environmental data and models as a condition for balancing uncertain scientific arguments against uncertain social, ethical, moral and legal arguments in managing environmental systems. A core challenge lies in minimizing all these uncertainties. The cost-effectiveness framework presented was developed to be used as a pragmatic support tool for policies analysis envisaged here in terms of alternative allocations of resources, the objective being to find the agro environmental measures that contribute most to achieving goals at minimum costs. Such an approach could help Basin Committees or Water Agencies better target implementation of measures and financial incentives to farmers where appropriate. A lack of cost-effectiveness is one of the major drawbacks of incentive schemes for mitigation measures and their availability is also restricted by a lack of funds. Based on CE approaches, the choice of a mitigation programme could be then a system of locally specified management incentives.

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