



# Cost-benefit analysis of nitrate abatement in the Souffel catchment (France): Sensitivity study of the damage and spatialization of the abatement effort

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3 **Cost-benefit analysis of nitrate abatement in the Souffel catchment**  
4 **(France): sensitivity study of the damage and spatialization of the**  
5 **abatement effort**

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## Abstract

Nitrate transfer from agricultural sources via river networks remains a serious unresolved and complex issue. This article proposes an economic analysis of the optimal reduction of this nitrate. A linear transformation and transport model of nitrogen inputs from agricultural sources in the form of nitrate from five agricultural areas towards a hydrographic network in France is used to calculate the optimal effort to reduce nitrogen inputs on the basis of a cost-benefit analysis (CBA). A sensitivity study is implemented with different damage scenarios. In addition, efforts to reduce uniform and spatialized inputs are compared. In particular, our results show the determining role of the magnitude of the damage. The ratio of 1 to 3 between the low and high range of its estimation would make it possible to attain good status, as specified by the Water Framework Directive (WFD), without having to resort to the exemption procedure, decreasing the average optimal nitrate concentration from 47 mg/l to 42 mg/l. Moreover, this would increase the absolute and relative benefits of spatialization by a factor of 9 and 2, respectively.

**Keywords:** cost-benefit analysis; nitrogen pollution; sensitivity analysis; environmental damage; abatement effort spatialization.

## 1. Introduction

The Water Framework Directive (WFD) of 23 October 2000 established an objective of good water status before 2015 for all of the water bodies in the European Union. However, the directive also included an exemption mechanism to permit the assignment of

a less stringent objective or to extend the timescales for achieving a particular objective in the event, for example, of disproportionate costs justified by a cost-benefit analysis (Article 4.4 of the WFD).

The principle of the CBA is simple. Scientific and technical information is combined for a set of competing options, plus information about the social preferences associated with the different options, in order to characterize the benefits and the costs (Boardman et al., 2001). Assessment should be presented in a form that allows a decision maker to rapidly comprehend the net effects of a project or policy (Hundloe et al., 1990). The comparison of the costs and benefits of the different possible options thus provides a relatively simple decision criterion: the option with the most advantageous cost-benefit ratio is chosen from among the options (Hanley and Spash, 1995), which also include the status quo solution.

During the first cycle of the WFD in France (ending in 2015), CBAs were carried out on 710 water bodies. Three-quarters of them revealed costs far outweighing the benefits (Feuillette et al., 2016), allowing an exemption of the time necessary to achieve the good water status objective for the following cycles (ending in 2021 and 2027). These results led to an examination of the methods used to assess the environmental benefits of an improvement in the quality of aquatic environments (Feuillette et al., 2016). First, it is necessary to identify the direct benefits (reduction in the cost of drinking water treatment, increase in fishing resources, etc.) as well as the indirect benefits (possibility of swimming, satisfaction of taking a walk or living next to a less polluted river, etc.) and, second, to assign a monetary value in order to compare environmental benefits to policy costs in the same unit of measurement. Many methods exist to do this, with methodological choices such as ecosystemic services and the impacted population to be taken into account, which may appear to be arbitrary. The results may therefore be very closely linked to the initial methodological choices.

This article examines the role of assumptions about the assessment of environmental benefits (or damage, depending on whether the nitrate concentration decreases or increases in the river) in determining optimal policies for reducing water pollution. The contribution of this article is twofold:

First, with respect to the CBA literature, the aim here is not to compare two policy options like in the literature but, instead, to build an agro-hydro-economic model to define the optimal abatement effort under different assumptions concerning the environmental benefit. This will thus shed light on the WFD's exemption policy.

Second, this article is in line with the literature on the spatialization of the abatement effort, which consists of requiring different abatement efforts depending on the location of the emissions and, more precisely, on the benefits of this spatialization according to the parameters of the model. For this purpose, optimal policies with the same abatement effort and with spatialized efforts will be compared. The opportunity to spatialize will thus be observed according to the different assumptions about the environmental benefits. In addition to making a contribution to the literature on spatialization, the results will be put into perspective with the policies of the French water agencies responsible for implementing the WFD.

The study area is the Souffel catchment (Bas-Rhin, Eastern France), a WFD water body of 132 km<sup>2</sup> in the Rhine-Meuse river basin in France. This water body has suffered from excessive pollution for many years, mainly due to nitrates from agricultural sources as well as pesticides. The CBA therefore focuses on the effort to reduce farm nitrogenous (N) inputs over the entire catchment in order to decrease the nitrate concentration in the surface waters of the Souffel catchment.

Section 2 of the article is devoted to a literature review of the CBAs and the spatialization of the abatement effort. In Section 3, “Materials and methods”, we describe the transformation and transport model for nitrogen fertilizers in the form of nitrate used to define the concentration of nitrate in each reach of the water body as a function of the level of N inputs in each agricultural area. The methods that made it possible to calculate the costs linked to the reduction of N inputs and the benefits linked to nitrate reduction in the water are also examined. Section 4, “Results”, presents the results of the CBAs for the eight scenarios according to the damage hypothesis and the uniform or spatialized effort. Finally, Section 5, "Discussion", provides an analysis of the results with respect to the literature on CBAs and spatialization, and a perspective on real environmental policies.

## **2. Literature review**

The cost-benefit analysis (CBA) is a key tool for evaluating welfare gains or losses from an investment (O’Mahony, 2021). It is a support tool for the implementation of projects and public policies (or private strategies) that has been used for many years now in North America, the United Kingdom and in the Scandinavian countries. As of 1936 in the United States, the Flood Control Act specified that navigation-improvement projects and flood-control projects on American waterways only be authorized if the benefits outweighed the estimated costs (Fuchs and Anderson, 1987). Since the end of the 1970s, the United States has extended the use of CBAs to the environment. As of this time, American law requires that all draft regulations with significant impacts be evaluated by a CBA (Treich, 2008). CBAs are also recommended by most of the major international organizations: World Health Organization, United Nations Environment Program, etc. (Destandau et al., 2020).

CBAAs have been used in some recent environmental studies to assess the advantages of a reforestation policy in China (Xian et al., 2020), a catchment restoration policy in Taiwan (Lin et al., 2020), to compare different seeding practices in Spain (Puértolas et al., 2012) and the United States (Turk et al., 2017), and to evaluate the advantages of replacing a conventional olive grove management system with an organic one in Italy (Sgroi et al., 2015). In these studies, CBAs are used to compare the costs and benefits of an ex-post policy (Xian et al., 2020; Lin et al., 2020), an ex-ante policy (Sgroi et al., 2015; WFD derogation procedure), or to compare the net benefit of different actions (Puértolas et al., 2012; Turk et al., 2017). To do this, the cost and benefit of a given action are calculated. Our study is unique in that it first defines the continuous functions of cost and benefit according to the level of effort necessary to reduce inputs in order to determine the optimal reduction effort of farm N inputs, i.e., the one that maximizes the net benefit.

Our study also examines two different options to reduce nitrogen inputs from agricultural sources: a uniform reduction over the whole catchment, or a spatialized reduction according to the specific agricultural area of the catchment. The terms of uniform and spatialized effort for the farmers will be used. The question of the spatialization of the abatement effort to increase the net benefit was dealt with in the literature as of 1974 (Tietenberg, 1974). In fact, the same quantity of inputs in the environment will not lead to the same concentration in the surface waters depending on the application site since the transformation, transport and subsequent dilution processes in the surface waters will vary within the catchment itself. We use the term “transfer coefficient” to indicate the share of a pollutant emitted at an emission point such as a farm plot, and then found at the level of a reception point within the river network. Moreover, the same concentration of a pollutant does not generate the same damage depending on the use of the impacted water

(production of drinking water, swimming water, biodiversity, various recreational uses, etc.). We refer to this as the “vulnerability” of the environment. Consequently, the location and the intensity of the effort to reduce inputs should depend on the marginal contribution to the damage, taking account of the transfer coefficients and the vulnerability of the reception sites, depending on the place of emission. However, this spatialization can also raise problems related to acceptability costs since the effort is not equally shared within a territory, or in terms of additional information to be obtained by the regulator to adjust the application of inputs and their fate before they reach the river network. As a result, it is crucial to know the factors that influence the magnitude of the advantage of spatializing the abatement effort on uniform regulation. O’Ryan (2006), Destandau and Nafi (2010), Krysiak and Schweitzer (2010) and Destandau (2013) all addressed this issue. It appears that it is the heterogeneity of the transfer coefficients, the marginal contributions to damage, the shape of the abatement cost curves and the shape of the environmental benefit curve in the event of a first-level regulation (maximization of the net benefit), or the intensity of the standard in the case of a second-level regulation (attaining an environmental standard at the lowest possible cost), which either justifies taking spatialization into account or not.

In this study, the results of the CBAs for uniform and spatialized regulations are analyzed according to different scenarios related to environmental benefits (or damage depending on whether the nitrate concentration decreases or increases in the river): magnitude of the total damage and vulnerability of the environment to nitrate. The term “uniform vulnerability” is used when the damage depends only on the nitrate concentration in the river, whereas the term “heterogeneous vulnerability” is used when the damage also depends on the



proximity of the riverside population. This is because the indirect negative effects are only perceived by the riverside population.

### **3. Materials and methods**

#### *3.1. Agro-hydrological approach*

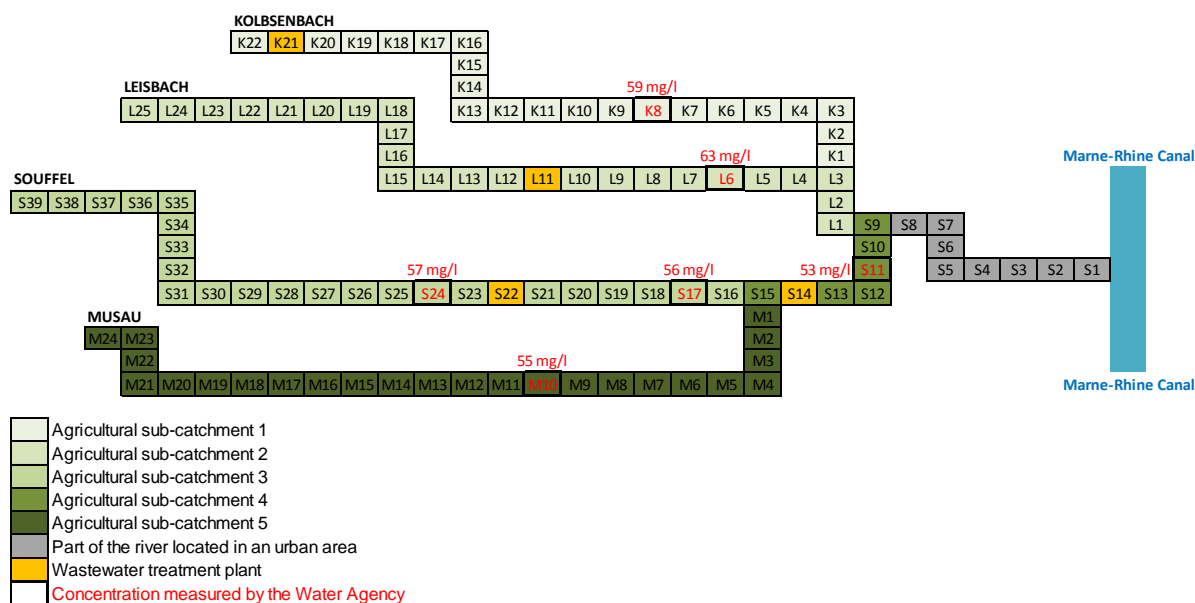
In order to make a cost-benefit analysis of the reduction N farm inputs for the purpose of decreasing the nitrate concentration in a catchment, a model of the input-concentration relationship is necessary. Models focus either, from a static point of view, on the impact of the reduction in fertilizer application, or, from a dynamic point of view, on the application schedule and the dynamic of nitrate transfers toward the river network (EU Sciences HUB, 2014). The input-concentration relationship is highly variable over time and closely linked to meteorological conditions and soil type: inputs are applied at specific periods during the year; the nitrogen surplus is leached in the fall under the plots and then flows into the rivers by horizontal flow from the catchments towards the river when it rains; and the conversion of the mass into nitrate concentration directly depends on the flow that also varies over time. Nevertheless, even if some models like SWAT (Soil and Water Assessment Tool) (Rasolomanana, 2012) attempt to take these temporal variations into account, the approach developed in this work to compare costs and benefits is based on average annual values, whether it be a question of river flows (Fig. A.1) or of the export of nitrate from the catchments to the river. This degree of simplification remains sufficiently realistic for the study catchment because the monthly and spatial variations in nitrate concentrations are low for the six Rhine-Meuse Water Agency (RMWA) stations, leading

to a homogeneity of the nitrate transfer within the catchment itself. The model that linearly links the quantity of nitrogen applied in the five agricultural areas to the nitrate mass that arrives in the catchment, broken down into 500-m homogeneous reaches, is defined below.

### *3.1.1. Souffel catchment*

The outlet of the Souffel catchment is located 3 km northwest of Strasbourg (Bas-Rhin, France; 48° 38' 20" N, 7° 44' 35" E). The catchment is flat (average slope:  $2.4 \pm 2.2\%$ ) with a total river network of 79 km. The crop types of this catchment are homogeneously distributed and constitute 84.5% of the total area (Corine Land Cover 2012; v18.5.1; <http://land.copernicus.eu>). The Souffel catchment encompasses four rivers: the Souffel and its two tributaries, the Leisbach and the Musau, plus the Kolbsenbach (a tributary of the Leisbach).

Figure 1 represents the hydrographic network discretized into a 500-m grid with its six RMWA monitoring stations for this catchment and the average associated nitrate concentrations that served as a reference for the WFD for the period 2015-2016-2017. It also includes the four discharges from the Wastewater Treatment Plants (WTP) and the connectivity of each section to one of the five agricultural areas considered.



**Fig. 1.** Souffel catchment schematized into 500-m reaches. Nitrate measured and sources of pollution. Each reach is designated by the initial of the river (K, L, S and M) where it is located and the number indicates its location in relation to the river outlet.

As illustrated on Fig. 1, the structure of the river network can be broken down into five agricultural areas, each one associated with a portion of the river. Since the industries in the catchment are connected to the WTPs, industrial pollution is automatically included in the urban pollution.

### 3.1.2. Nitrate from urban and natural sources

The average mass of nitrogen discharged every day by the four WTPs is provided by the Rhine-Meuse Water Agency on the basis of self-monitored data on its website (eau-rhin-meuse.fr). For the year 2017, these nitrogen quantities were: 0.9 kg/day (in K21); 4.92 kg/day (in L11); 25.99 kg/day (in S22); and 6.26 kg/day (in S14). These quantities are converted into equivalent nitrate to obtain nitrate flows in mg/s (Sumelius et al., 2005). The nitrate flow discharged by these WTPs is then transferred upstream from reach to reach, undergoing a reduction via the self-purifying capacity of the environment. Based on

the expertise of the Rhine-Meuse Water Agency for river networks in Eastern France, a reduction of 5% (value used by the Rhine-Meuse Water Agency within the framework of the Pegase model (Everbeck et al., 2019)) of the nitrate mass every 500 m was adopted. (Fig. A.1) According to Verchère (2010), the nitrate present in the surface waters also has natural sources linked to the nitrogen cycle, estimated at between 3 mg/l to 5mg/l. Based on this study, a contribution of 4 mg/l is associated with this geochemical background, making it possible to determine the anthropic contribution from RMWA concentrations (Fig. A.2) by calculating the difference.

### *3.1.3. Nitrate from agricultural sources*

Nitrogen pollution from agricultural sources therefore corresponds, within the limit of the uncertainties related to estimates of the natural geochemical background and urban discharges, to the difference between the nitrate concentrations measured on Figs. 1 and A.2. The transformation and transfer process of nitrogen in the form of nitrate from each agricultural area towards each reach of the river depends on the quantity of nitrogen  $N_i$  applied each year in each agricultural area  $i$ , of the share  $\pi_i$  (zone diffusion coefficient) of  $N_i$  that we generally find in the form of nitrate in the rivers, and, for each sub-catchment, of the share  $\rho_{ij}$  of  $\pi_i N_i$  that is exported towards each of the reaches  $j$  of  $i$ .

In 2017, farmers applied an average of 177 kg/ha/year of nitrogen on the Souffel catchment (Martin and Baudet, 2017). This value will be considered homogeneous for the whole Souffel catchment to calculate the parameter  $N_i$ . The total annual quantity of nitrogen applied is obtained by multiplying  $N_i$  by the surface area of each agricultural area (Table 1).  $\rho_{ij}$  is defined as a function of the share of agricultural land use around the reach.  $\pi_i$  is defined so as to find the concentration targeted by the monitoring stations. This latter

parameter thus incorporates the uncertainties related to nitrate sources, flows and degradation coefficients in the river network.

**Table 1**

Quantity of nitrogen applied every year for each agricultural area.

| Agricultural sub-catchment | Agricultural surface (ha) | Nitrogen applied (kg/year) |
|----------------------------|---------------------------|----------------------------|
| Kolbsenbach (SC1)          | 3 244                     | 574 188                    |
| Leisbach (SC2)             | 1 831                     | 324 087                    |
| upstream-Souffel (SC3)     | 2 715                     | 480 555                    |
| downstream-Souffel (SC4)   | 735                       | 130 095                    |
| Musau (SC5)                | 1 853                     | 327 981                    |

To define  $\rho_{ij}$ , the coefficient indicating the intensity of agricultural land use around reach  $\sigma_{ij}$  is determined beforehand (Fig. B.1). The allocation key  $\rho_{ij}$  of the nitrate from each agricultural area  $i$  in each of the reaches  $j$  of  $i$  (Fig. B.2) is calculated as follows:

$$\rho_{ij} = \frac{\sigma_{ij}}{\sum_{j=1}^{J_i} \sigma_{ij}}$$

where  $J_i$  is the number of reaches in the agricultural sub-catchment  $i$ .

By designating  $C_{ij}$  the estimated concentration of nitrate from agricultural sources in the river at the level of the monitoring stations (difference between the nitrate concentrations observed and the estimated concentrations of urban and natural sources in Figs. 1 and A.2), and  $\delta_{ij}$  the flow in the  $j^{th}$  reach of the agricultural sub-catchment  $i$  (Fig. A.1), Equation 1 is proposed:

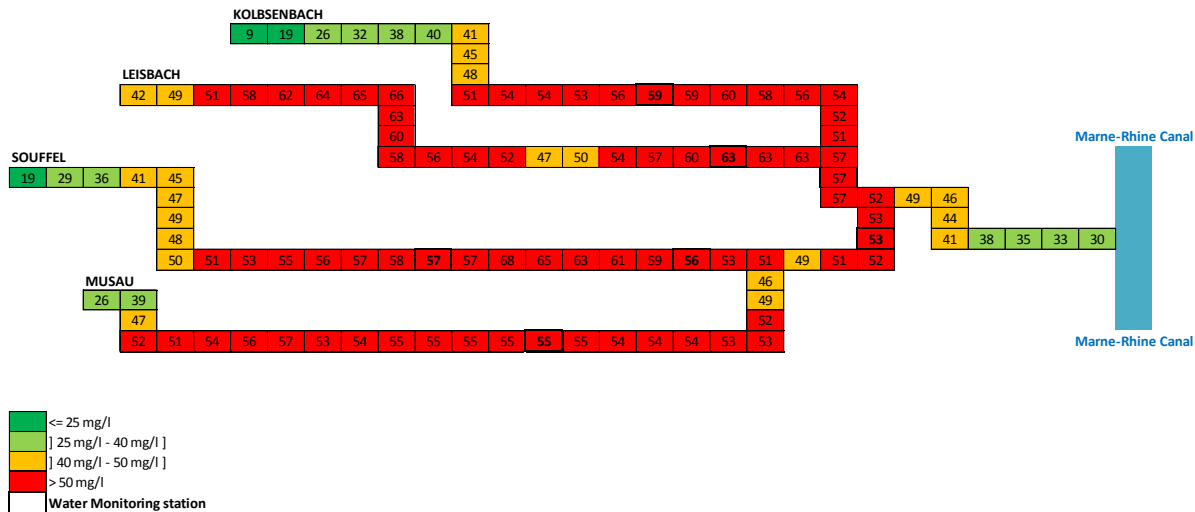
$$C_{ij} = \frac{N_i \cdot \pi_i \cdot \rho_{ij} + 95\% \cdot N_i \cdot \pi_i \cdot \rho_{i(j-1)} + (95\%)^2 \cdot N_i \cdot \pi_i \cdot \rho_{i(j-2)} + \dots}{\delta_{ij}} \quad (1)$$

This makes it possible to define the five zone diffusion coefficients  $\{\pi_1, \pi_2, \pi_3, \pi_4, \pi_5\}$  using the concentrations observed at the RMWA monitoring stations:

$$\{0.7\%, 2.1\%, 1.4\%, 2.3\%, 1.2\%\} \quad (2)$$

“0.7%” for  $\pi_1$  signifies that for 100 kg of farm N applied in agricultural sub-catchment 1, there will be 0.7 kg of nitrate in the river.

Given the quantities of nitrogen applied in each agricultural area (Table 1), the allocation keys for each reach (Fig. B.2), and the five zone diffusion coefficients, concentrations of nitrate from agricultural sources for all of the catchment reaches can be deduced using Equation 1. The total nitrate concentration for all of the catchment reaches (Fig. 2) can then be estimated after adding the concentrations from urban and natural sources (Fig. A.2).



**Fig. 2.** Estimated nitrate concentration (mg/l) for all of the reaches of the Souffel catchment.

The above model thus makes it possible to recalculate the nitrate concentrations for all of the reaches, regardless of the quantities of nitrogen inputs applied in each agricultural area.

### 3.2. Economic model

The economic model aims at estimating the total ecological damage generated by nitrate on the Souffel catchment as a function of the nitrate level in the different river reaches or

the ecological benefits of a reduction in nitrate concentration. It also estimates the cost of a reduction in farm N inputs for each agricultural area via a reduction in yield.

### *3.1.1. Costs linked to the reduction of nitrogen inputs*

Farm N inputs make it possible to increase crop yield. Consequently, the cost of a reduction in these inputs can be calculated by the decrease in crop yield. This estimation method has been implemented for over 30 years, especially in the literature from the Scandinavian countries. Within the framework of the Helsinki Convention, a reduction target of 50% of the nutrients was fixed to reduce eutrophication in the Baltic Sea. This target led to numerous studies, like Gren (1991) on Laholm Bay in Sweden, and Helin (2006) in Southwestern Finland.

Our aim was not to estimate the decrease in yield for a specific N input reduction target but, instead, to obtain a continuous function that would allow us to estimate the impact on yield at every level of input reduction. Crop yield as a function of input quantity obviously depends on the crop, the type of fertilizer (mineral, organic), the proportions in which the input is applied and weather conditions. To develop and apply the CBA method, we adopted the scenario of a homogeneous crop like corn that would cover 80% of the agricultural area of the Souffel catchment. In Canada, the University of Guelph (Niemeyer, 2020) estimated the evolution of corn yield according to the quantity of nitrogen inputs over several years. This study, as well as a study on actual fertilization practices and corn yields for the Souffel catchment (Martin and Baudet, 2017), made it possible to estimate the average yield of a hectare of corn as a function of the quantity of nitrogen fertilizer applied (see Table C.1), and to then predict the loss of benefit or cost, depending on the reduction of nitrogen inputs, as follows (see Fig. C.1):

$$\text{“Cost of reduction”} = 0.0452 \times (\text{“N reduction”})^2 + 0.5535 \times (\text{“N reduction”}) \quad (3)$$

This equation corresponds to a cost for the farmer and not to the cost of the policy that aims at requiring or encouraging the farmer to implement this reduction. Moreover, only the short-term reduction of inputs is considered without any structural modification of the crops, e.g., rotation of corn with other crops, modifications of cropping techniques, etc.

### *3.1.2. Benefits linked to the reduction of nitrate in the water*

Estimating the ecological damage generated by pollution is a tricky business that may involve highly varied scenarios from one study to another. Which impacts from pollution should be taken into account? How can we assign a monetary value to these impacts? In a first step, we define a total range of damage generated by the nitrate over the entire Souffel catchment, on the basis of the existing literature. This damage is then broken down for each reach with, successively, a scenario of uniform vulnerability followed by one of heterogeneous vulnerability.

#### *3.1.2.1. Magnitude of damage on the whole Souffel catchment*

Some studies estimate the damage generated by nitrates of agricultural origin on a large scale, like Pretty et al. (2002) for England and Wales, or Dodds et al. (2008) for the US. Damage is linked to the impairment of use values, like the additional cost of producing drinking water, or non-use values like the recreational uses of water. In France, the estimations of the OECD (2012) or Aquamoney (2010) are good examples. At a smaller scale, other studies on water bodies focused on the impact of nitrates under conditions of high vulnerability, like the pollution of groundwater coupled with other pollutants, e.g., pesticides, or an impact on health due to the use of polluted water to produce drinking water (Rinaudo et al., 2005; Khader et al., 2013). The damage proposed in these studies



was normalized to obtain a range of damage per inhabitant, and converted into “2020 euros” in order to be used in this study on the Souffel. The damage was then multiplied by the population of the Souffel catchment (61,248 inhabitants). The range of damage  $D$  obtained, taken from the highly different contexts of the studies cited above, was analyzed to prioritize the results in order to determine those that overestimate and those that underestimate the damage on the Souffel (see Table D.1).

After this phase of prioritization, the total damage  $D$  on the Souffel was estimated on the basis of a low estimation of €1,000,000/year and a high estimation of €3,000,000/year.

#### 3.1.2.2. Damage per reach: uniform vulnerability

The total damage  $D$  estimated above for the whole catchment therefore represents the damage generated by the nitrate currently present in all of the catchment reaches. The link between damage and nitrate concentration is essential to calculate the damage generated for each reach.

Even though it imposes a concentration threshold of 50 mg/l, Article 6 of the Nitrate Directive (91/676/EC) requires monitoring when the concentration exceeds 25 mg/l, the lower threshold for damage generated (Brink et al., 2011; Ward et al., 2005). Therefore, in the uniform vulnerability scenario, damage is assumed to depend only on the nitrate concentration, which increases at a rate  $d_U$  above 25 mg/l. Consequently, if  $C_j$  is the concentration for each of the 110 reaches  $j$ , we can write:

$$\sum_{j=1}^{110} d_U \cdot \gamma_j = D \quad \Leftrightarrow \quad d_U = \frac{D}{\sum_{j=1}^{110} \gamma_j} \quad (4)$$

where:  $\gamma_j = 0$  if  $C_j \leq 25$   
 $\gamma_j = (C_j - 25)$  if  $C_j > 25$

By fixing a value for  $D$  and by incorporating the concentration values in Fig. 2 into Equation (4), the value  $d_U$  for each reach can be obtained. With this coefficient, we can recalculate the total damage when the concentrations evolve.

### 3.1.2.3. Damage per reach: heterogeneous vulnerability

In contrast with the uniform vulnerability scenario, it is possible to make the damage conditional on the specific water use (swimming water, water to produce drinking water, fish water, etc.) or non-use (stopping or abandoning walks if the river has been eutrophicated due to an excess of nitrate, etc.) that will depend on the riverside population. With this in mind, the damage generated by a certain concentration on a section of the river will be assumed to correlated with the number of riverside inhabitants living within 5 km of the river:  $p_j$  (see Fig. E.1). The marginal damage  $d_H$  is calculated as follows:

$$\sum_{j=1}^{110} d_H \cdot p_j \cdot \gamma_j = D \quad \Leftrightarrow \quad d_H = \frac{D}{\sum_{j=1}^{110} p_j \cdot \gamma_j} \quad (5)$$

Avec: 
$$\begin{aligned} \gamma_j &= 0 & \text{si } C_j \leq 25 \\ \gamma_j &= (C_j - 25) & \text{si } C_j > 25 \end{aligned}$$

## 4. Results

The agro-hydrological and economic models developed made it possible to vary the levels of nitrogen from agricultural sources applied on the five agricultural areas, and to calculate the total associated input reduction cost, as well as the damage generated by the residual nitrate after reduction, or the benefit generated by the reduction of this nitrate, depending on the different damage scenarios: magnitude for total damage  $D$  and uniform or

heterogeneous vulnerability. The CBAs were initially carried out assuming the same reduction in nitrogen inputs (uniform effort over the whole catchment), and then considering a spatialized reduction from one agricultural area to another. The advantage in terms of cost-benefit of this spatialization is then discussed. Eight nitrate concentration profiles were thus obtained for the Souffel catchment by maximizing the net benefit (ecological benefit minus the abatement cost) according to these scenarios (Appendix F), and are detailed in the following sections.

#### *4.1. CBAs for a uniform nitrogen reduction on the Souffel catchment*

The high scenario of current damage  $D$  on the Souffel catchment of 3M euros rather than the low scenario of 1M results in an increased effort for farmers since nitrogen inputs are reduced by 12% rather than by 2% (Table 2).

**Table 2**

Results of the CBAs for a uniform nitrogen reduction, considering uniform or heterogeneous vulnerability with a high and low damage range  $D$ .

|                          | Uniform Vulnerability |         | Heterogeneous Vulnerability |         |
|--------------------------|-----------------------|---------|-----------------------------|---------|
|                          | 1M                    | 3M      | 1M                          | 3M      |
| Input reduction (effort) | 1,9%                  | 12,6%   | 2,0%                        | 12,7%   |
| Cost (€/year)            | 24 975                | 361 547 | 25 393                      | 367 370 |
| Benefit (€/year)         | 146 162               | 944 545 | 146 905                     | 954 241 |
| Net benefit (€/year)     | 121 187               | 582 998 | 121 512                     | 586 871 |

This leads to an increase in the cost as well as in the benefits in order to arrive at a net benefit that is almost five times greater. The result is logical, as illustrated in Fig. E.2. When the impact on the environment appears to be low, it may seem to be too costly to reduce fertilizer inputs compared to the damage avoided, thus generating a smaller

pollution effort. We can therefore imagine that the high level of current pollution on this catchment can be explained by an under-evaluation of the damage caused by nitrate in this region. This result is also the consequence of our scenarios that target the reduction of nitrogen inputs alone. A modification of cultural practices not taken into account in this study could lower the cost of reducing N inputs, and the intensity of the damage linked to nitrate could be re-evaluated on the basis of new studies or as the result of improved awareness of the medium and long-term consequences.

Heterogeneous vulnerability does not modify the results obtained with a homogenous vulnerability on the totality of the rivers (Table 2). In fact, the difference is not significant in light of the uncertainties. The supplementary effort to fine-tune the scenarios for this catchment in terms of damage generated by including the proximity of the riverside population does not contribute any additional value in the CBA. This result illustrates that a fine-tuning of the vulnerability scenarios in this specific case was not particularly useful. However, this result can obviously not be generalized. It will depend on the spatial distribution of emission zones, the different flow levels and the vulnerability to pollution.

#### *4.2. CBAs for a spatialized reduction of nitrogen depending on the agricultural area*

As discussed previously, the same quantity of applied nitrogen will not necessarily generate the same environmental damage, depending on the agricultural area where the emission takes place. In fact, the fate and transfer of nitrogen in the form of nitrate from the plots to the river depends on the biogeochemical cycle of the nitrogen at the water-soil-plant interface and the variable water transfers over space and the course of the year. Consequently, the diffusion coefficient used in this approach to incorporate the totality of these processes that determine the transformation and the transfer of applied nitrogen

towards the rivers in the form of nitrate may vary depending on the agricultural area. It is what we have referred to as the diffusion coefficient (result (2)), which encompasses the entire set of agronomic, chemical and hydrological processes, from the nitrogen applied to the nitrate in the rivers. The hydrological functioning of each reach will then provide a different dilution potential depending on the flow range considered that determines the observed concentration. (Fig. A.1). Finally, each reach may present a specific vulnerability that is taken into account in this study by the heterogeneous vulnerability scenario, which makes the same level of nitrate concentration more or less harmful depending on its assigned use and the perception of the riverside population considered (Fig. E.1). For example, this is the case for localized recreational uses or even the fact of enjoying the view of a clean river or suffering from the view of one that is degraded by excessive eutrophication within proximity of one's home. As a result, this spatial variability of the fate, transfer and impact of the nitrate can justify spatialized efforts (see Table 3).

**Table 3**

Results of the CBAs for a spatialized nitrogen reduction, considering uniform or heterogeneous vulnerability with a high and low damage range  $D$ .

|                          | Uniform Vulnerability |           | Heterogeneous Vulnerability |         |
|--------------------------|-----------------------|-----------|-----------------------------|---------|
|                          | 1M                    | 3M        | 1M                          | 3M      |
| Kolbsenbach (SC1)        | 0%                    | 7%        | 1%                          | 9%      |
| Leisbach (SC2)           | 5%                    | 22%       | 4%                          | 19%     |
| upstream-Souffel (SC3)   | 2%                    | 12%       | 2%                          | 12%     |
| downstream-Souffel (SC4) | 0%                    | 2%        | 1%                          | 10%     |
| Musau (SC5)              | 4%                    | 19%       | 3%                          | 16%     |
| Cost (€/year)            | 32 420                | 420 920   | 27 908                      | 390 104 |
| Benefit (€/year)         | 159 925               | 1 063 293 | 151 936                     | 999 710 |
| Net benefit (€/year)     | 127 505               | 642 373   | 124 028                     | 609 606 |

By permitting a spatialized effort that depends on the agricultural area, there will be potential “winners” and “losers”. The farmers in the areas of the Leisbach and the Musau

will have to make greater efforts, whereas those of the Kolbsenbach and of the downstream Souffel will have to make less and perhaps none at all. The additional effort on the Leisbach can be explained by a high diffusion coefficient: 2.1% (result (2)), compared to the other areas (the difference in the diffusion coefficient can be explained by the pedology, the average distance between plots, etc.). It signifies that for 100 kg of nitrogen applied in this area, 2.1 kg of nitrate are found in the water. Consequently, reducing the nitrogen upstream will have a greater impact on the environment. For the area of the Musau, the additional effort can be explained by a relatively weak flow (Fig. A.1). The average flow on this river is 30 l/s, compared to almost 40 l/s, 50 l/s, 60 l/s, 180 l/s on the Kolsenbach, the Leisbach, the upstream Souffel and the downstream Souffel, respectively. The dilution effect is therefore lower on the Musau, meaning that the same mass of nitrate from the upstream agricultural areas will produce a higher concentration. Inversely, reducing the nitrogen upstream will reduce the nitrate concentration even more on this river. In contrast, the lesser effort can be explained by a very low diffusion coefficient on the Kolbsenbach: 0.7%, and a very high average flow on the downstream Souffel (Fig. 2). These “transformation/transport” effects within the catchments and the “dilution” in the reaches from upstream to downstream therefore explain, regardless of the damage scenario, that the effort will be greater on the Leisbach River, followed by the Musau River, and then the upstream Souffel River. Compared to a uniform reduction effort (Table 2), spatializing reduction efforts does not significantly increase the differences between the results of the CBAs obtained for uniform or heterogeneous vulnerability scenarios (Table 3). The effort between sub-catchments is slightly more balanced for the heterogeneous vulnerability scenario. This can be explained by the fact that vulnerability linked to weaker uses downstream due to a stronger dilution capacity is compensated for

by a greater vulnerability of non-uses linked to a higher riverside population downstream (Fig. B.1).

#### 4.3. Advantage of spatializing the fertilization reduction effort

As the theory predicts (Tietenberg, 1974), the net benefit is higher for a spatialized effort (Tables 2 and 3). To measure the advantage of spatializing the fertilization reduction effort according to the damage scenarios, an “absolute advantage” is first calculated by establishing the difference between the net benefit of the spatialized effort and the uniform effort. A “relative advantage” is then obtained by dividing this difference by the uniform effort (Table 4). The relative advantage thus expresses the percent of increase of the net benefit when we switch from a uniform effort to a spatialized effort.

**Table 4**

Advantage of spatializing the fertilization reduction effort.

|   | Uniform Vulnerability |        | Heterogeneous Vulnerability |        |
|---|-----------------------|--------|-----------------------------|--------|
|   | 1M                    | 3M     | 1M                          | 3M     |
| Absolute advantage of spatialization (€/year) | 6 318                 | 59 375 | 2 516                       | 22 735 |
| Relative advantage of spatialization          | 5%                    | 10%    | 2%                          | 4%     |

We can observe that in Table 4, when the magnitude of damage increases from 1M to 3M, the absolute advantage of spatialization is multiplied by almost 10, and the relative advantage by two.

These results are consistent with those in the literature (Krysiak and Schweitzer, 2010; Destandau, 2013) that indicate that a higher degree of damage requires a greater spatialization of the input reduction effort or of decontamination, depending on the study.

The absolute and relative advantages of spatialization are approximately 2.5 times greater for uniform vulnerability than for heterogeneous vulnerability (Table 4). As previously mentioned, this can be explained by the fact that in the case of heterogeneous vulnerability, the “dilution” effect is compensated by a “population” effect that makes marginal contributions to the damage less heterogeneous (Table 3).

## **5. Discussion**

In this section, we discuss the results obtained in the previous section on the use of CBAs to reduce water pollution and, particularly, non-point source pollution, as well as on the appropriateness of spatializing the abatement effort.

### *5.1. CBAs for the optimal reduction of non-point source pollution*

CBAs are used to rationalize policy choices by comparing the financial or monetary consequences of different options. In the literature, we can cite studies that compare two possible options at the local level such as the choice of sowing with native or conventional herbaceous species (Turk et al., 2017), or the opportunity of converting to organic farming in the case of olive groves (Sgroi et al., 2015). We can also mention the net benefit of more comprehensive policies compared to a status quo situation such as the "Grain for Green" reforestation program in 1999 in China (Xian et al., 2020), or the Watershed Conservation and Restoration Program in 2017-2018 in Taiwan (Lin et al., 2020). Similarly, the CBA of the WFD’s exemption policy only compares two policy options: achieving good status or status quo, but does not use the CBA to determine the optimal policy. This is what we



propose in this paper thanks to an agro-hydro-economic model, with, in addition, a sensitivity study depending on the different damage assumptions.

Concerning nitrates, the WFD defines a standard of 50 mg/l for water bodies if they are to have good status in terms of this parameter. In France, this standard is reached when the 50-mg/l concentration is not exceeded for more than 10% of the measurements. Thus, in Table 5 below, we are interested in the number and percentage of reaches on the Souffel where the optimal solution generates values that exceed this 50 mg/l standard for our eight scenarios.

**Table 5**

Reaches below the 50 mg/l standard (a total of 110 reaches).

| N Reduction | Vulnerability | Total damage | Number of reaches (>50mg/l) | Share of reaches (>50mg/l) |
|-------------|---------------|--------------|-----------------------------|----------------------------|
| uniform     | uniform       | low          | 41                          | 37%                        |
| uniform     | uniform       | high         | 13                          | 12%                        |
| uniform     | heterogeneous | low          | 41                          | 37%                        |
| uniform     | heterogeneous | high         | 13                          | 12%                        |
| spatialized | uniform       | low          | 37                          | 34%                        |
| spatialized | uniform       | high         | 9                           | 8%                         |
| spatialized | heterogeneous | low          | 36                          | 33%                        |
| spatialized | heterogeneous | high         | 8                           | 7%                         |

We can observe that good status is achieved, or almost, for the high range of the damage estimate (3M), but only for 1/3 of the reaches with a nitrate concentration above 50 mg/l for the low range (1M). This means that the WFD exemption would be allowed for a low damage estimate, with pollution that could remain at the current level, i.e., 74 of the reaches with concentrations above 50 mg/l, or 2/3 of the reaches (Fig. 2), whereas the abatement effort should be made with the high estimate.

In the WFD, the benefits are often estimated by transferring values from other studies carried out in other places and adapted to the area in question. Indeed, estimating the

environmental benefits specific to a given area is a long and costly task. It is a method similar to the transfer of values that allowed us to define the low and high ranges that we used in this study. We must therefore acknowledge the risk of referring to value transfers and, more generally, to basing a policy on the monetization of environmental benefits. In our case, we can see that it is all the more tempting for a local decision-maker under pressure to use the low range of damages because of the considerable consequences on the abatement costs. In Table 2, we can see that by multiplying the damage by 3, the abatement effort is multiplied by 6 and the abatement cost by 14. The environmental results are therefore very sensitive to the assumptions of a method that has inherent assessment difficulties (Browne and Ryan, 2011).

The next step to achieve good status of water bodies in Europe is 2027. Our study shows that in the event of a exemption request, particular consideration should be given to the way in which the environmental benefits have been calculated. Moreover, we can imagine that new impacts of pollutants may be observed over time, validating the choice of using high ranges. An ambitious environmental policy will thus have to be based on the high range of damage. Finally, rather than allowing a status quo in the event of an exemption, the assessment of an optimal pollution level should be considered so that a subsequent abatement effort is made, even if it does not make it possible to obtain good status.

## *5.2. Advantage of spatializing the abatement effort*

According to Tietenberg (1974), when the marginal contribution to the damage of polluting emissions differs according to the place of emission, the abatement effort must be spatialized. The marginal contribution to the damage consists of the transfer coefficients (transformation of the emissions from the emission sources into pollutant concentration at

the receptors) and the vulnerability of the receptor sites (magnitude of the damage generated according to the concentration) when the policy aims at maximizing the net environmental benefit of the abatement costs (first-best), and of the transfer coefficients and the intensity of the quality standard when the policy aims at achieving an ambient standard at the minimum abatement cost (second-best). However, the benefit of spatialization is context-dependent. Indeed, Tietenberg (2006) presents the results of empirical studies on air pollution reduction policies in the United States where it appears that the gains in pollution control costs to reach the same standard are reduced, depending on the study, by 6.6% to 95.7% with a spatialized vs. uniform policy. Several authors have studied the contextual parameters that could influence the benefit of spatialization (O'Ryan, 2006; Destandau and Nafi, 2010; Krysiak and Schweitzer, 2010; Destandau, 2013).

Our paper contributes to this literature by measuring the benefit of spatializing the abatement effort under different assumptions about the damage generated by nitrate pollution, including the magnitude of the damage at a low range (1M) and a high range (3M). This makes it possible to study the benefits of spatialization according to the policy issue.

In a theoretical study, Destandau and Nafi (2010) show that the advantage of spatialization decreases when the standard is more restrictive in the case where the marginal costs of cleanup are convex. In fact, for very stringent standards, the pollution control costs are so high that the relative difference between uniform and spatialized regulation becomes negligible. On the other hand, for marginal linear abatement costs, the standard has no impact on the benefit of spatialization. In another theoretical study, Destandau (2013) once again attempted to identify the impact of the environmental policy issue on the advantage of spatialization, but this time in a first-best regulation where one seeks to maximize the net benefit. This time, it appeared that the magnitude of the damage could increase the

589 advantage of spatialization. The results obtained in first-best and second-best scenarios  
590 thus seem to be contradictory, since when the issue of abatement (more restrictive  
591 standard, greater damage) has an effect on the advantage of spatialization, it reduces it in  
592 one case and increases it in the other. The author justifies this by the fact that in the case of  
593 the second-best scenario, damage from pollution and, therefore, a greater abatement  
594 benefit, makes it possible to reach a better quality level, whereas for the second-best  
595 scenario, the same quality standard will be reached for the uniform and spatialized  
596 regulation. In our study, for example, the average pollution on the Souffel catchment is 47  
597 mg/l at the optimum for the low damage range vs. 42 mg/l for the high range (Appendix  
598 F).

599 Our work contributes to this literature by being the first empirical study to examine the  
600 spatialization effect in terms of the first-best regulation. Our results first show that  
601 spatialization increases net profits by at least 2% and, at most, 10% (Table 4). This is  
602 relatively low compared to the studies cited by Tietenberg (2006). This can be explained  
603 by the limited size of the study catchment, and probably reveals relatively homogeneous  
604 characteristics of the polluters in terms of their marginal contribution to the damage. We  
605 can also observe in Table 3 that, consistent with the theoretical results of Destandau  
606 (2013), the benefit of spatialization increases with the damage generated by the pollution.  
607 More precisely, we can see that for a total damage multiplied by 3, the absolute advantage  
608 is multiplied by 9, and the relative advantage by 2.

609 Another contribution of our work compared to the literature is that the transfer coefficient  
610 is divided into two parts: the diffusion coefficient, which illustrates the transport effect,  
611 and the flow rate, which illustrates the dilution effect. This allows a more detailed analysis  
612 of the results. We observed that for spatialized efforts, an additional effort is required on  
613 the Leisbach (Table 3) due to a high dilution coefficient, and on the Musau due to a low

614 flow rate. In contrast, the smaller effort on the Kolbsenbach and on the downstream-  
615 Souffel is explained by a low diffusion coefficient and a high flow rate, respectively. We  
616 also point out (Table 4) that spatialization is less attractive for heterogeneous vulnerability,  
617 with an absolute and relative advantage that is 2.5 times higher for uniform vulnerability.  
618 This is due to the fact that the high population density is located downstream where the  
619 flow rate is very high. Thus, in the case of heterogeneous vulnerability, the greater impact  
620 due to the population density (vulnerability effect) is compensated for by the lower impact  
621 due to the high flow rate (dilution effect). The marginal contribution to the damage is thus  
622 less heterogeneous than in the case of the uniform vulnerability hypothesis.

623 In France, the WFD is implemented by the water agencies, which collect fees from  
624 polluters and pay subsidies for pollution control. Each water agency chooses a basic fee  
625 rate that applies to different pollutants and that can define areas within its catchment where  
626 rates will be higher or lower. For example, the rates will be lower for off-shore ocean  
627 disposal, or for organic matter in very rainy areas due to the dilution effect, and higher in  
628 more vulnerable areas: popular tourist destinations along the coast, areas vulnerable to  
629 eutrophication for nitrogen- and phosphorus-based substances, groundwater, etc. The  
630 existing literature and our results show that when the pollution control stakes are higher, a  
631 lower spatialization can be justified if the objective is to reach an ambient standard. On the  
632 other hand, if the aim is to maximize the net environmental benefit, a higher spatialization  
633 may be favored. For the water agencies, the choice of a greater spatialization will thus have  
634 to be linked to the choice between strict compliance with the good status of the WFD, or a  
635 greater commitment to the restoration of aquatic environments.

## 6. Conclusion

Calculation of the optimal reduction of a pollutant in the natural environment is directly linked to the means considered to reduce effluents and to assumptions concerning the calculation of the environmental benefit of this reduction. This article targeted the optimal applied N fertilizer reduction effort, leading to the reduction of nitrate in the rivers of an urban catchment where corn is the major crop. Two types of reductions were considered: a uniform fertilizer reduction effort with the same reduction proportion in each of the five sub-catchments identified, and a spatialized reduction involving different efforts depending on the sub-catchment. As for the environmental effort linked to the decrease of nitrate in the water, two magnitudes were compared (damage 1M and 3M), as well as two vulnerability scenarios: uniform vulnerability in which the environmental damage depends only on the nitrate concentration, and heterogeneous vulnerability in which the damage also depends on the proximity of the riverside population.

The magnitude of damage and, consequently, the perception of negative effects generated by the presence of nitrate, has a very big impact on the intensity of the reduction of N fertilizers necessary to reach optimal concentrations. Actually, for a damage multiplied by 3, the optimal abatement effort is multiplied by 6, and the abatement costs by 14. Moreover, the optimal average nitrate concentration in the basin decreases from 47 mg/l to 42 mg/l, which would allow the good status of the WFD to be achieved without an exemption. In addition, the shift from the low to the high damage range also increases the benefit of spatializing the abatement effort by a ratio of 1 to 9 for the absolute benefit, and from 1 to 2 for the relative benefit. In the case of spatialization, an additional abatement effort will be required for farmers whose effluents have a higher diffusion coefficient or whose effluents drain into reaches of the river with weaker flows. Our results also show

that the benefit of spatialization is decreased by a factor of 2.5 when it is considered that the damage depends not only on the nitrate concentration but also on the population density. This is due to the fact that the population density effect, downstream of the catchment, is counterbalanced by a dilution effect resulting from a higher flow. However, this result is context-specific. A higher vulnerability upstream of the catchment due to a fishing or bathing area would increase the benefit of spatialization. Basing environmental policies on the assessment of benefits thus deserves careful consideration to maintain ambitious objectives. We can cite the example of the WFD, which aims to achieve good status of water bodies in Europe by 2027 at the latest. At the national and regional level, the authorities responsible for implementing the WFD will also have to examine the possibility of spatializing the abatement effort according to the homogeneity or heterogeneity of the diffusion coefficients and the vulnerability of certain areas to pollution, as well as according to the intensity of the effort required from the polluters.

This study mainly focused on scenarios related to damage and the spatialization of the fertilizer reduction effort. Future studies could perhaps target different scenarios linked to the cost of reducing nitrogen fertilizers. For example, rather than simply reducing the quantity of nitrogen fertilizers, a change in agricultural practices could be considered. Crop rotations that include legumes, in particular, could lead to a significant reduction in the use of fertilizers. These rotations could also make it possible to decrease the use of pesticides that impact the same water bodies, and could generate other benefits in terms of biodiversity, for example. Likewise, we can ask ourselves if the cost of policies aimed at modifying behaviors and practices should be included in decontamination costs as well. These reflections would also make it possible to examine the opportunity of spatializing the abatement effort as a function of the magnitude and heterogeneity of the cleanup and policy costs. Finally, the approach proposed can be further enriched if the required data are

derived from the diffusion coefficients used on the basis of physically-grounded agro-hydrological models, making it possible to incorporate a greater share of the complexity of the fate of nitrogen fertilizers in the water-soil-plant continuum.

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## 810 **List of abbreviations and symbols**

811

### 812 **Abbreviations:**

813 CBA: Cost-Benefit Analysis

814 N input: Nitrogenous input

815 RMWA: Rhine-Meuse Water Agency

816 SWAT: Soil and Water Assessment Tool

817 WFD: Water Framework Directive

818 WTP: Wastewater Treatment Plant

819

### 820 **Symbols:**

821  $C_j$ : Estimated concentration of nitrate from agricultural sources in reach  $j$

822  $D$ : Total damage on the Souffel

823  $d_U$ : Damage rate per unit concentration above 25 mg/l in the uniform vulnerability scenario

824  $d_H$ : Damage rate per unit concentration above 25 mg/l in the heterogeneous vulnerability

825 scenario

826  $J_i$ : Number of reaches in the agricultural sub-catchment  $i$

827  $N_i$ : Quantity of nitrogen applied each year in each agricultural area  $i$

828  $\gamma_j$ : Number of nitrate concentration units above 25 mg/l

829  $\delta_{ij}$ : Flow in the  $j^{th}$  reach of the agricultural sub-catchment  $i$

830  $\pi_i$ : Diffusion coefficient = share of  $N_i$  found in the form of nitrate in the rivers

831  $\rho_{ij}$ : Share of  $\pi_i N_i$  that is exported towards each of the reaches  $j$  of agricultural area  $i$

832  $\sigma_{ij}$ : Intensity of agricultural land use around the reach  $j$  of agricultural area  $i$

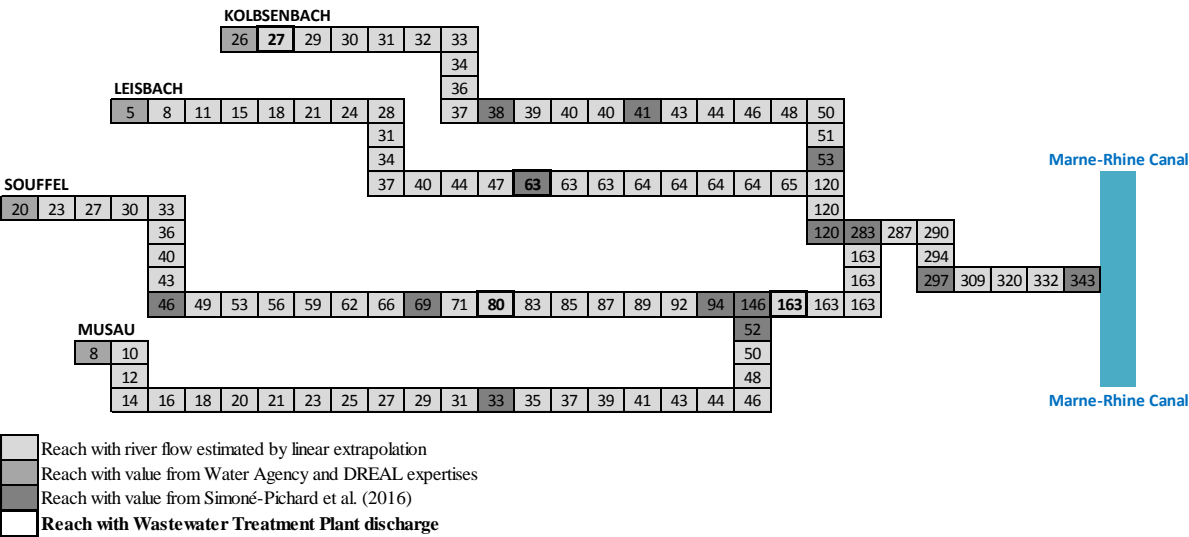
833

Appendices

Appendix A

Figure A.1

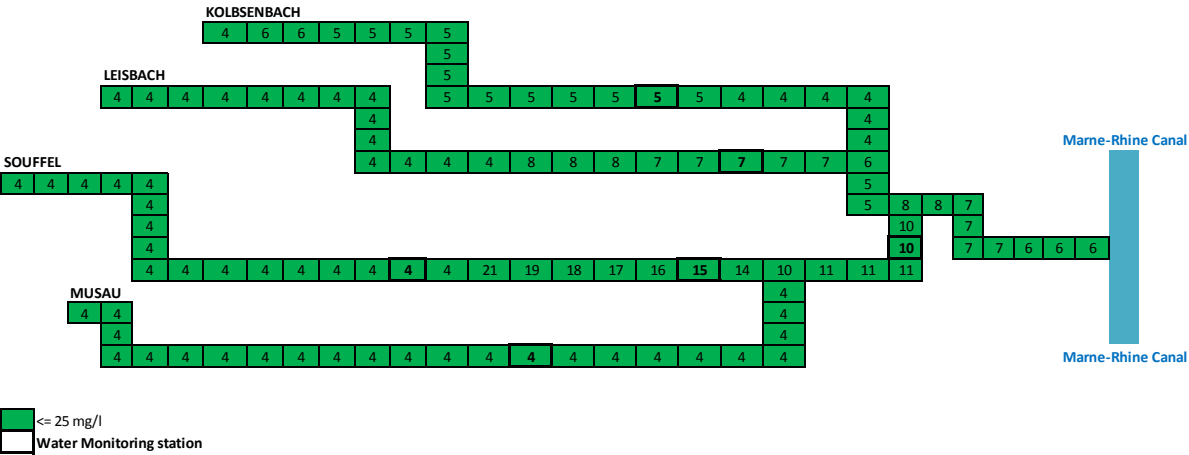
Estimated flows for each catchment reach (l/s).



**Legend:** Our estimation of the average flow for each reach was based on several sources: Simoné-Pichard et al. (2016) who estimated the average flow for 14 catchment reaches; expertises of the “DREAL Grand Est” (Regional Direction of the Environment, Development and Housing for the Grand-Est region); the Rhine-Meuse Water Agency that allowed us to determine the average flows at the river sources; and data on water treatment plants (WTPs) in order to estimate the average added flow discharged from the WTP. For reaches whose flow is still unknown, the estimation is supplemented by a linear extrapolation between the known upstream and downstream flows.

Figure A.2

Estimated nitrate concentrations (mg/l) from urban and natural sources.



Appendix B

Figure B.1

Coefficient indicating the intensity of agricultural land use around the reach.

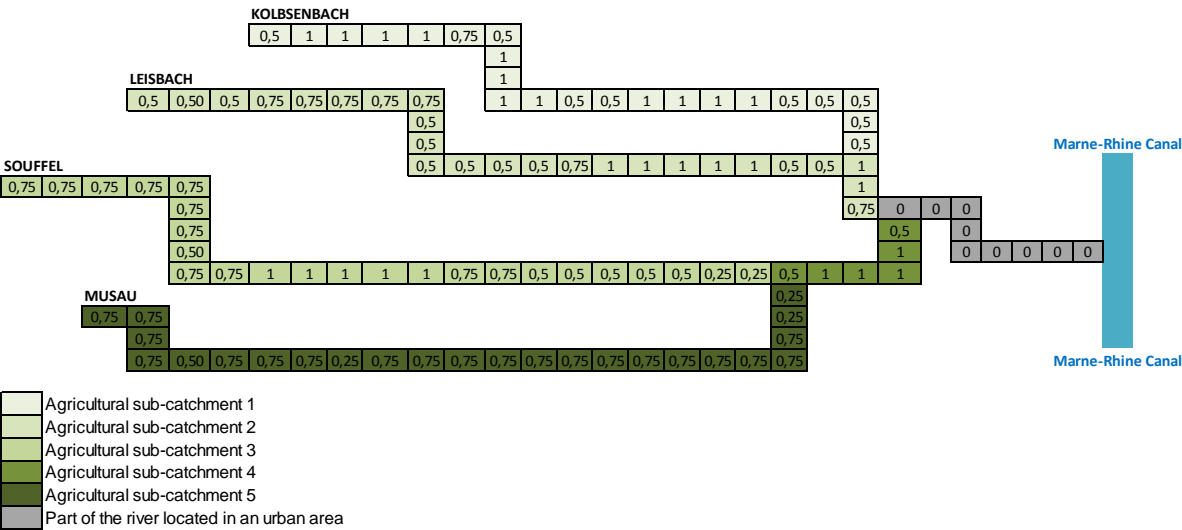
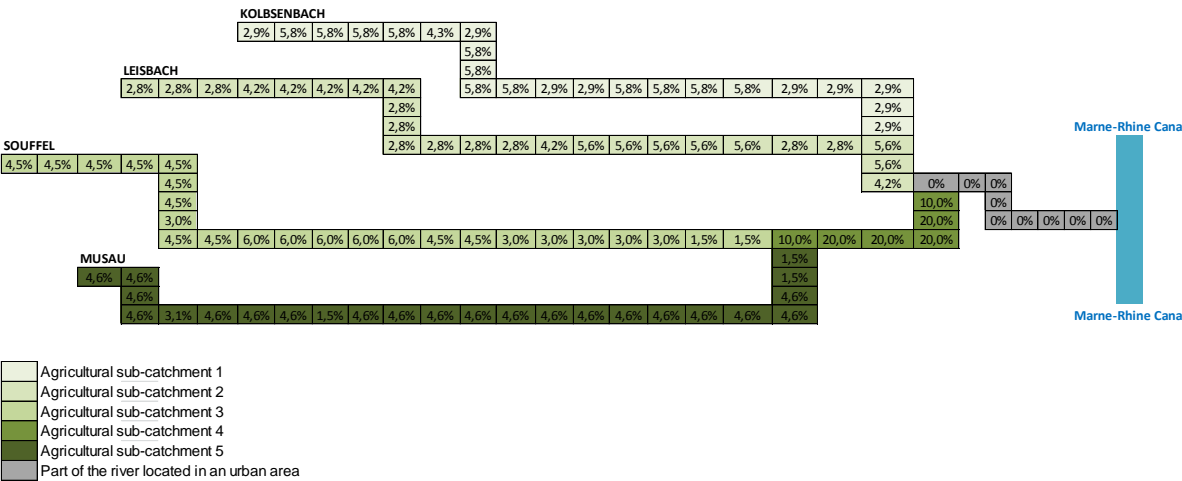


Figure B.2

Nitrate allocation key for each agricultural sub-catchment in each of the reaches.



## Appendix C

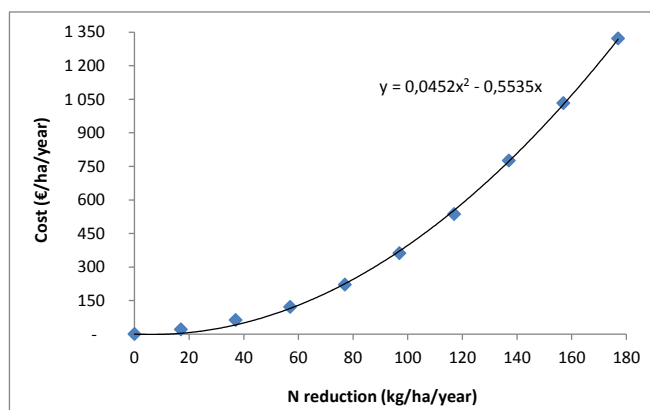
**Table C.1**

Construction of the cost of reducing farm N inputs.

| N application (kg/ha) | Corn Production (kg/ha) | Corn Sales (€/ha) | Cost N (€/ha) | Benefit (€/ha) | N reduction (kg/ha) | Cost of reduction (€/ha) |
|-----------------------|-------------------------|-------------------|---------------|----------------|---------------------|--------------------------|
| 0                     | 2 400                   | 396               | -             | 396            | 177                 | 1 322                    |
| 20                    | 4 300                   | 710               | 24            | 686            | 157                 | 1 033                    |
| 40                    | 6 000                   | 990               | 48            | 942            | 137                 | 776                      |
| 60                    | 7 600                   | 1 254             | 72            | 1 182          | 117                 | 536                      |
| 80                    | 8 800                   | 1 452             | 96            | 1 356          | 97                  | 362                      |
| 100                   | 9 800                   | 1 617             | 120           | 1 497          | 77                  | 221                      |
| 120                   | 10 550                  | 1 741             | 144           | 1 597          | 57                  | 121                      |
| 140                   | 11 050                  | 1 823             | 168           | 1 655          | 37                  | 63                       |
| 160                   | 11 450                  | 1 889             | 192           | 1 697          | 17                  | 21                       |
| 177                   | 11 700                  | 1 931             | 212           | 1 718          | 0                   | -                        |

**Figure C.1**

Cost according to the reduction of N inputs.



**Legend:** As we can see in Table C.1, the variation in corn yield linked to its sales price (0.165€/kg according to the Grand Est Farm Observatory, 2020) makes it possible to obtain “Corn Sales” according to the level of applied N input. By subtracting the cost of this input (1.2€/kg according to the Alsace Chamber of Agriculture), we obtain the “Benefit”. The reduction of this benefit according to the reduction of the input, “N reduction”, will be our “Cost of reduction” (Fig. C.1).

Using polynomial regression, we obtain for each hectare:

$$\text{“Cost of reduction”} = 0.0452 \times (\text{“N reduction”})^2 + 0.5535 \times (\text{“N reduction”})$$

## Appendix D

**Table D.1**

Analysis of the literature concerning damage generated by farm N inputs.

| Author                   | Context                  | Low range<br>(€/year) | Single estimate<br>(€/year) | High range<br>(€/year) | Comments  |
|--------------------------|--------------------------|-----------------------|-----------------------------|------------------------|---|
| Pretty et al. (2002)     | England and Wales        |                       | 203 343                     |                        | Not all the population is concerned with pollution. Dividing by the total population underestimate our damage   |
| Dodds et al. (2008)      | U.S.A.                   |                       | 489 984                     |                        | Not all the population is concerned with pollution. Dividing by the total population underestimate our damage   |
| OECD (2012)              | France                   | 61 248                |                             | 979 968                | Not all the population is concerned with pollution. Dividing by the total population underestimate our damage   |
| Rhine-Meuse Water Agency | France                   |                       | 2 082 432                   |                        | Overestimation because not only nitrate, but, in the same time, Underestimation because only the damage over 50mg/l   |
| Aquamoney (2010)         | France                   | 551 232               |                             | 5 879 808              | Low range too low (nitrate concentration higher in the Souffel than in the national average) but High range too high (Souffel is a small catchment without tourism and drinking water production) |
| Rinaudo et al. (2005)    | Groundwater in France    |                       | 29 889 024                  |                        | Nitrate and pesticide pollution for groundwater used for drinking water production. This overestimates our damage.  |
| Khader et al. (2013)     | Groundwater in Palestine |                       | 41 709 888                  |                        | Nitrate damages on drinking water production and human health. This overestimates our damage.   |

**Legend:** The literature underestimates (green) or overestimates (red) the damage on the Souffel. Yellow means that it has not yet been determined.

For example, damage for a country, divided by all of its inhabitants, many of whom are not concerned by nitrate pollution, underestimates the damage on the Souffel, whereas other studies that estimate nitrate damage in groundwater or on the production of drinking water, contexts that are much more vulnerable than that of the Souffel catchment, overestimate the damage. We can also recall that the Rhine-Meuse Water Agency, within the framework of the cost-benefit analysis of the WFD, estimated the benefit that attaining good water status would generate on the Souffel catchment (i.e., the damage that would be eliminated). On the one hand, this value overestimates the damage because it does not just take account of nitrates, but, on the other hand, it underestimates it because it ignores the damage linked to a nitrate concentration that exceeds the threshold value of 50 mg/l.



Appendix E

Figure E.1

Inhabitants located in spots less than 5 km from each catchment section.

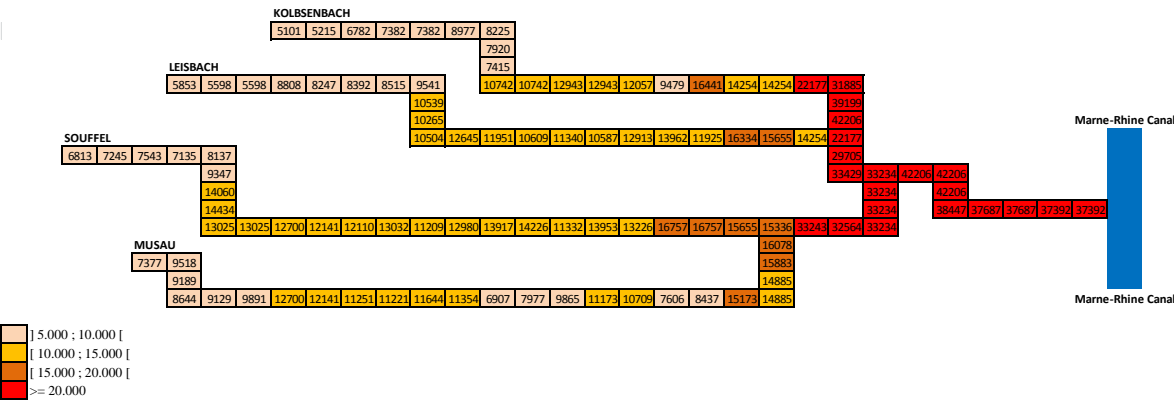
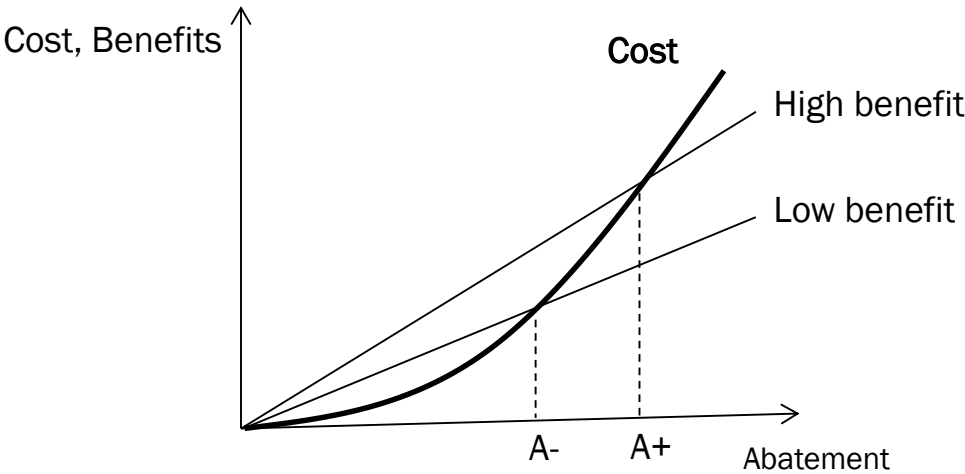


Figure E.2

Impact of the variation of the magnitude of the environmental benefit (or damage).



Appendix F

Optimal nitrate concentrations on the Souffel for each reach.

Figure F.1

Reduction of the UNIFORM effort, UNIFORM vulnerability and total damage of for the 1M scenario.

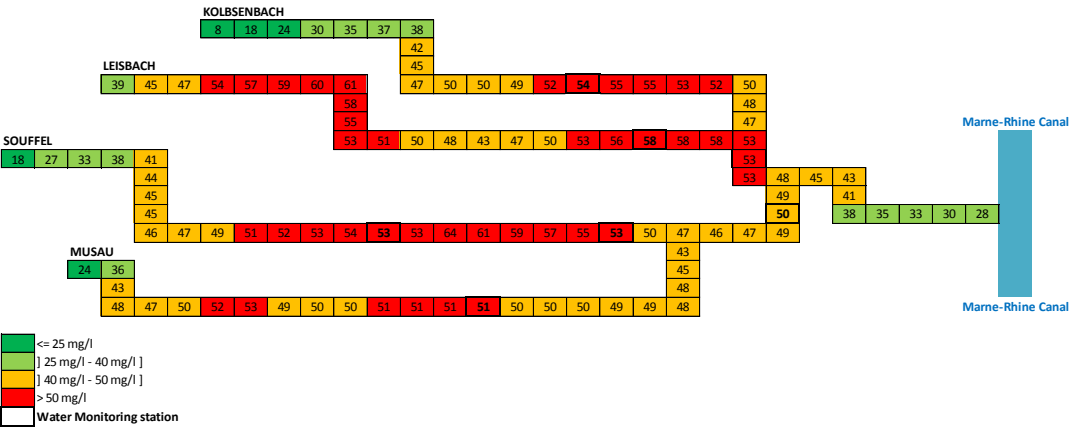
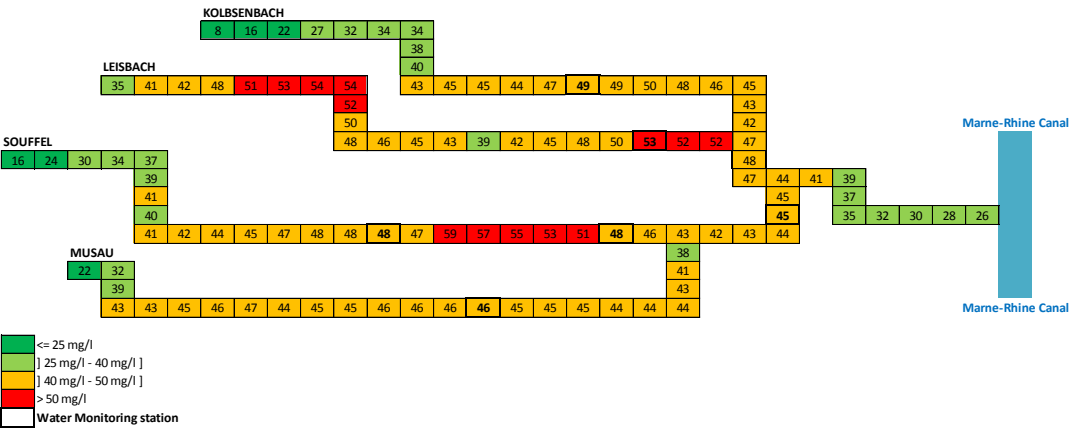


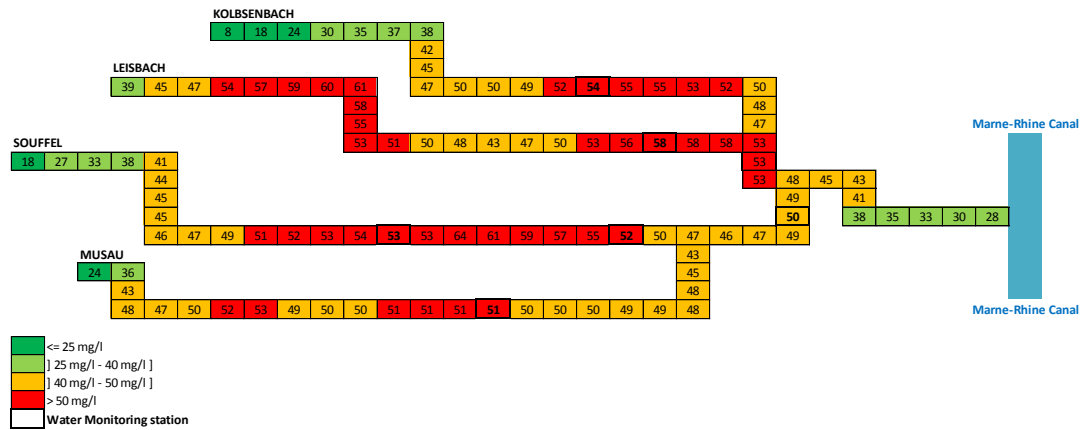
Figure F.2

Reduction of the UNIFORM effort, UNIFORM vulnerability and total damage for the 3M scenario.



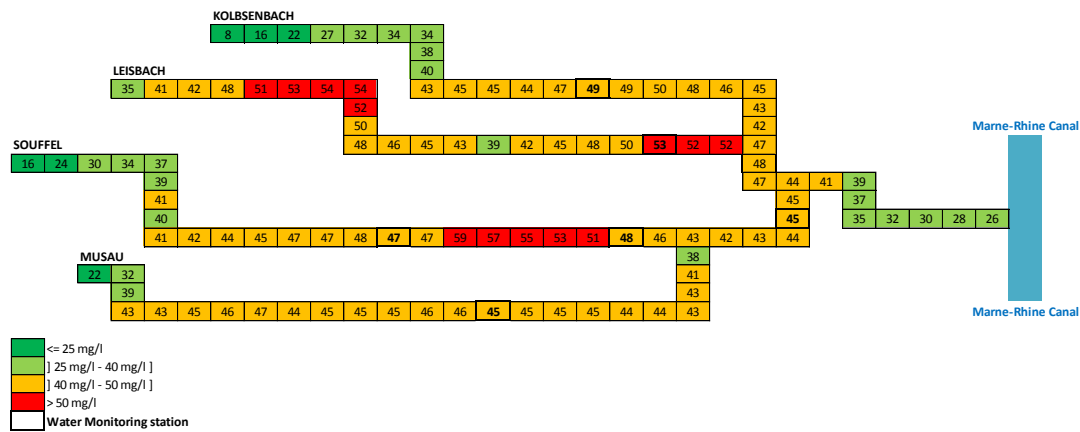
Reduction of the UNIFORM effort, HETEROGENEOUS vulnerability and total damage for the 1M scenario.

Reduction of the UNIFORM effort, HETEROGENEOUS vulnerability and total damage for the 1M scenario.



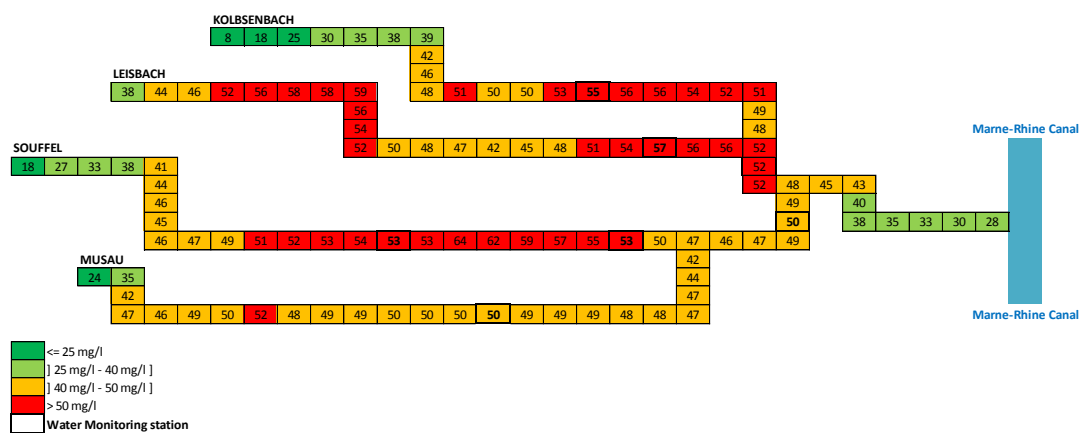
Reduction of the UNIFORM effort, HETEROGENEOUS vulnerability and total damage for the 3M scenario.

Reduction of the UNIFORM effort, HETEROGENEOUS vulnerability and total damage for the 3M scenario.

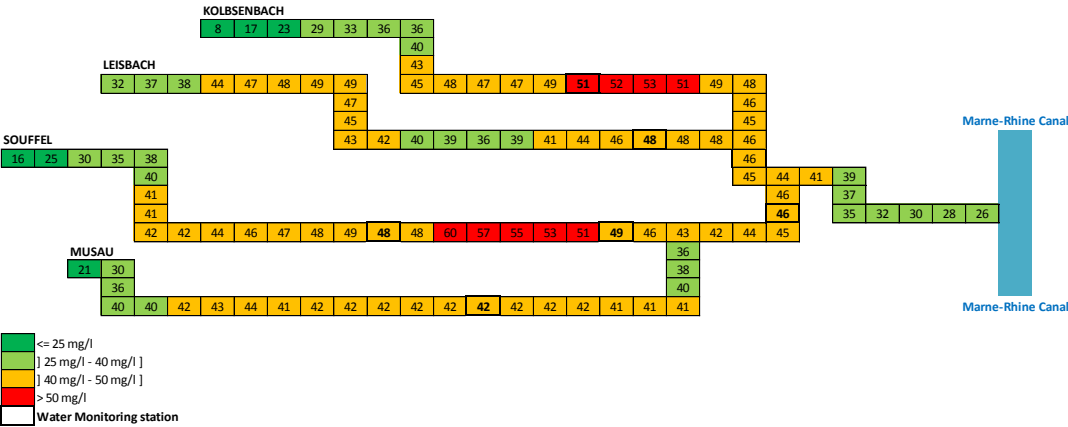


Reduction of the SPATIALIZED effort, UNIFORM vulnerability and total damage for the 1M scenario.

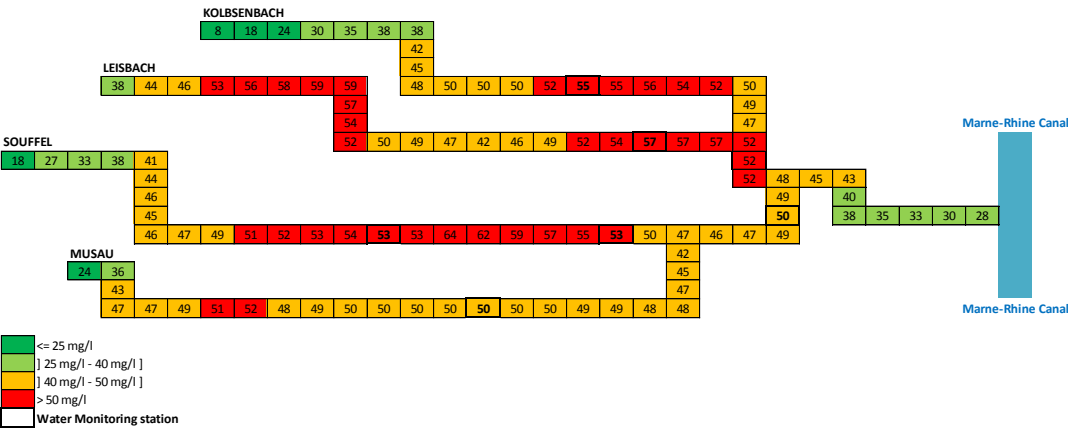
Reduction of the SPATIALIZED effort, UNIFORM vulnerability and total damage for the 1M scenario.



**Figure F.6**  
Reduction of the SPATIALIZED effort, UNIFORM vulnerability and total damage for the 3M scenario.



**Figure F.7**  
Reduction of the SPATIALIZED effort, HETEROGENEOUS vulnerability and total damage for the 1M scenario.



**Figure F.8**  
Reduction of the SPATIALIZED effort, HETEROGENEOUS vulnerability and total damage for the 3M scenario.

