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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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Youssef ZAITER

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10

11 **Abstract**

12

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

26

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

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30

31 **1. Introduction**

32

33 The Water Framework Directive (WFD) of 23 October 2000 established an objective
34 of good water status before 2015 for all of the water bodies in the European Union.

35 However, the directive also included an exemption mechanism to permit the assignment of

36 a less stringent objective or to extend the timescales for achieving a particular objective in
37 the event, for example, of disproportionate costs justified by a cost-benefit analysis (Article
38 4.4 of the WFD).

39 The principle of the CBA is simple. Scientific and technical information is combined for a
40 set of competing options, plus information about the social preferences associated with the
41 different options, in order to characterize the benefits and the costs (Boardman et al.,
42 2001). Assessment should be presented in a form that allows a decision maker to rapidly
43 comprehend the net effects of a project or policy (Hundloe et al., 1990). The comparison of
44 the costs and benefits of the different possible options thus provides a relatively simple
45 decision criterion: the option with the most advantageous cost-benefit ratio is chosen from
46 among the options (Hanley and Spash, 1995), which also include the status quo solution.

47 During the first cycle of the WFD in France (ending in 2015), CBAs were carried out on
48 710 water bodies. Three-quarters of them revealed costs far outweighing the benefits
49 (Feuillette et al., 2016), allowing an exemption of the time necessary to achieve the good
50 water status objective for the following cycles (ending in 2021 and 2027). These results led
51 to an examination of the methods used to assess the environmental benefits of an
52 improvement in the quality of aquatic environments (Feuillette et al., 2016). First, it is
53 necessary to identify the direct benefits (reduction in the cost of drinking water treatment,
54 increase in fishing resources, etc.) as well as the indirect benefits (possibility of swimming,
55 satisfaction of taking a walk or living next to a less polluted river, etc.) and, second, to
56 assign a monetary value in order to compare environmental benefits to policy costs in the
57 same unit of measurement. Many methods exist to do this, with methodological choices
58 such as ecosystemic services and the impacted population to be taken into account, which
59 may appear to be arbitrary. The results may therefore be very closely linked to the initial
60 methodological choices.

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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 km² 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.

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

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98 **2. Literature review**

99

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

111 CBAs have been used in some recent environmental studies to assess the advantages of a
112 reforestation policy in China (Xian et al., 2020), a catchment restoration policy in Taiwan
113 (Lin et al., 2020), to compare different seeding practices in Spain (Puértolas et al., 2012)
114 and the United States (Turk et al., 2017), and to evaluate the advantages of replacing a
115 conventional olive grove management system with an organic one in Italy (Sgroi et al.,
116 2015). In these studies, CBAs are used to compare the costs and benefits of an ex-post
117 policy (Xian et al., 2020; Lin et al., 2020), an ex-ante policy (Sgroi et al., 2015; WFD
118 derogation procedure), or to compare the net benefit of different actions (Puértolas et al.,
119 2012; Turk et al., 2017). To do this, the cost and benefit of a given action are calculated.
120 Our study is unique in that it first defines the continuous functions of cost and benefit
121 according to the level of effort necessary to reduce inputs in order to determine the optimal
122 reduction effort of farm N inputs, i.e., the one that maximizes the net benefit.

123
124 Our study also examines two different options to reduce nitrogen inputs from agricultural
125 sources: a uniform reduction over the whole catchment, or a spatialized reduction
126 according to the specific agricultural area of the catchment. The terms of uniform and
127 spatialized effort for the farmers will be used. The question of the spatialization of the
128 abatement effort to increase the net benefit was dealt with in the literature as of 1974
129 (Tietenberg, 1974). In fact, the same quantity of inputs in the environment will not lead to
130 the same concentration in the surface waters depending on the application site since the
131 transformation, transport and subsequent dilution processes in the surface waters will vary
132 within the catchment itself. We use the term “transfer coefficient” to indicate the share of a
133 pollutant emitted at an emission point such as a farm plot, and then found at the level of a
134 reception point within the river network. Moreover, the same concentration of a pollutant
135 does not generate the same damage depending on the use of the impacted water

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

153

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

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

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163

164 **3. Materials and methods**

165

166 *3.1. Agro-hydrological approach*

167

168 In order to make a cost-benefit analysis of the reduction N farm inputs for the purpose of
169 decreasing the nitrate concentration in a catchment, a model of the input-concentration
170 relationship is necessary. Models focus either, from a static point of view, on the impact of
171 the reduction in fertilizer application, or, from a dynamic point of view, on the application
172 schedule and the dynamic of nitrate transfers toward the river network (EU Sciences HUB,
173 2014). The input-concentration relationship is highly variable over time and closely linked
174 to meteorological conditions and soil type: inputs are applied at specific periods during the
175 year; the nitrogen surplus is leached in the fall under the plots and then flows into the
176 rivers by horizontal flow from the catchments towards the river when it rains; and the
177 conversion of the mass into nitrate concentration directly depends on the flow that also
178 varies over time. Nevertheless, even if some models like SWAT (Soil and Water
179 Assessment Tool) (Rasolomanana, 2012) attempt to take these temporal variations into
180 account, the approach developed in this work to compare costs and benefits is based on
181 average annual values, whether it be a question of river flows (Fig. A.1) or of the export of
182 nitrate from the catchments to the river. This degree of simplification remains sufficiently
183 realistic for the study catchment because the monthly and spatial variations in nitrate
184 concentrations are low for the six Rhine-Meuse Water Agency (RMWA) stations, leading

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

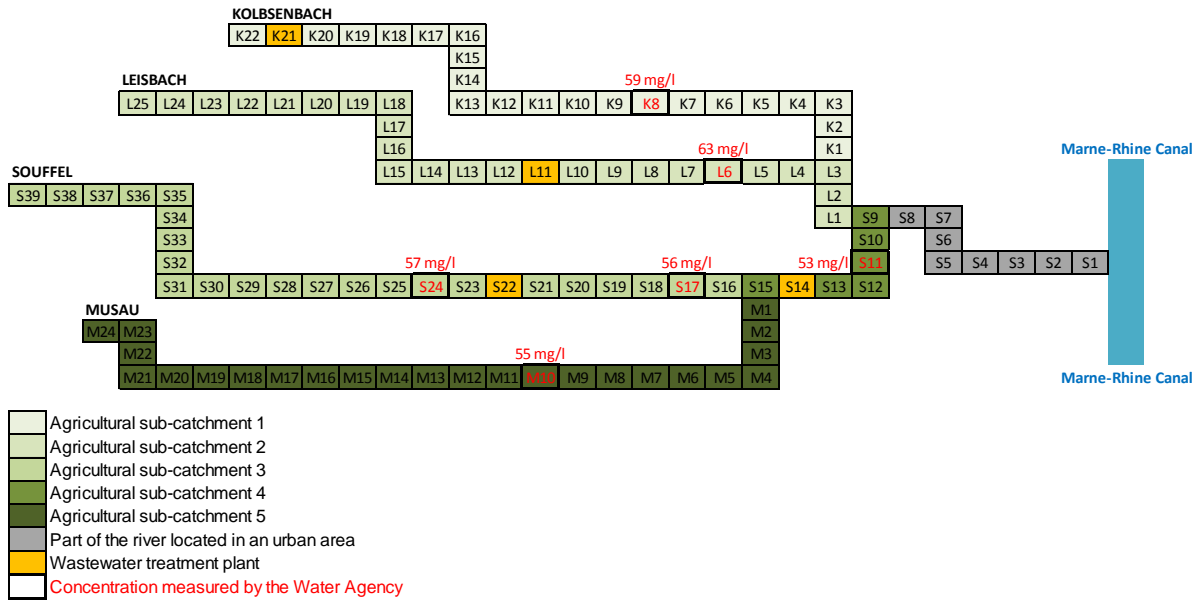
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189 *3.1.1. Souffel catchment*

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

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

202



203

204

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.

207

208

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.

212

213

3.1.2. Nitrate from urban and natural sources

214

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

220

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

229

230 *3.1.3. Nitrate from agricultural sources*

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

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

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

247

248 **Table 1**

249 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

250

251

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

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

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

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

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

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

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

264

265 “0.7%” for π_1 signifies that for 100 kg of farm N applied in agricultural sub-catchment 1,
 266 there will be 0.7 kg of nitrate in the river.

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

273

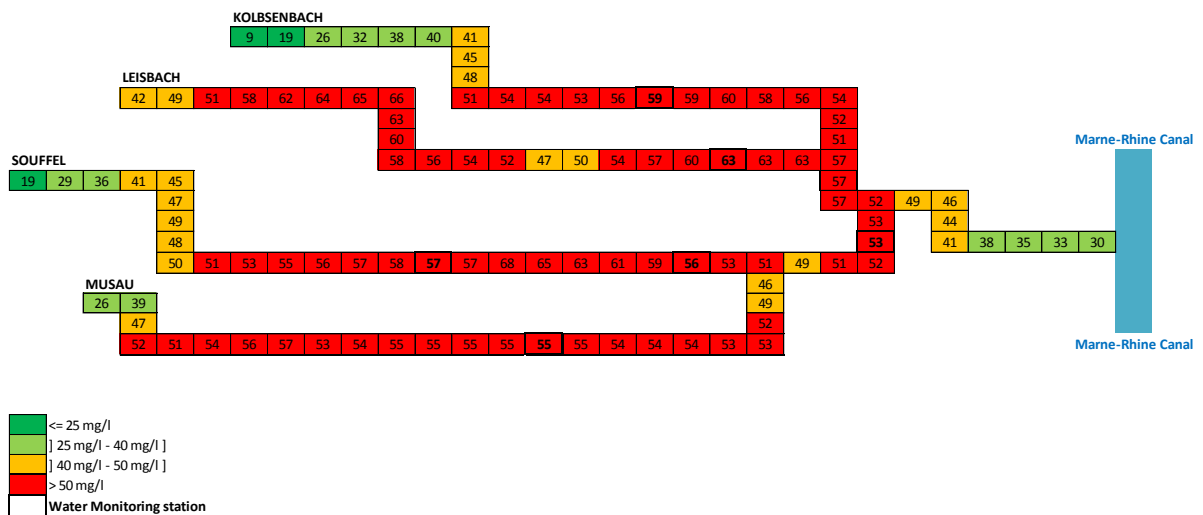


Fig. 2. Estimated nitrate concentration (mg/l) for all of the reaches of the Souffel catchment.

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

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

286

287 *3.1.1. Costs linked to the reduction of nitrogen inputs*

288 Farm N inputs make it possible to increase crop yield. Consequently, the cost of a
289 reduction in these inputs can be calculated by the decrease in crop yield. This estimation
290 method has been implemented for over 30 years, especially in the literature from the
291 Scandinavian countries. Within the framework of the Helsinki Convention, a reduction
292 target of 50% of the nutrients was fixed to reduce eutrophication in the Baltic Sea. This
293 target led to numerous studies, like Gren (1991) on Laholm Bay in Sweden, and Helin
294 (2006) in Southwestern Finland.

295 Our aim was not to estimate the decrease in yield for a specific N input reduction target
296 but, instead, to obtain a continuous function that would allow us to estimate the impact on
297 yield at every level of input reduction. Crop yield as a function of input quantity obviously
298 depends on the crop, the type of fertilizer (mineral, organic), the proportions in which the
299 input is applied and weather conditions. To develop and apply the CBA method, we
300 adopted the scenario of a homogeneous crop like corn that would cover 80% of the
301 agricultural area of the Souffel catchment. In Canada, the University of Guelph (Niemeyer,
302 2020) estimated the evolution of corn yield according to the quantity of nitrogen inputs
303 over several years. This study, as well as a study on actual fertilization practices and corn
304 yields for the Souffel catchment (Martin and Baudet, 2017), made it possible to estimate
305 the average yield of a hectare of corn as a function of the quantity of nitrogen fertilizer
306 applied (see Table C.1), and to then predict the loss of benefit or cost, depending on the
307 reduction of nitrogen inputs, as follows (see Fig. C.1):

$$308 \quad \text{“Cost of reduction”} = 0.0452 \times (\text{“N reduction”})^2 + 0.5535 \times (\text{“N reduction”}) \quad (3)$$

309

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

314

315 *3.1.2. Benefits linked to the reduction of nitrate in the water*

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

323

324 *3.1.2.1. Magnitude of damage on the whole Souffel catchment*

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

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

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

342

343 3.1.2.2. *Damage per reach: uniform vulnerability*

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

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

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

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

355

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

359

360 *3.1.2.3. Damage per reach: heterogeneous vulnerability*

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

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

368 Avec: $\gamma_j = 0 \quad \text{si } C_j \leq 25$
 $\gamma_j = (C_j - 25) \text{ si } C_j > 25$

369

370

371 **4. Results**

372

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

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

385

386 *4.1. CBAs for a uniform nitrogen reduction on the Souffel catchment*

387

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

391

392 **Table 2**

393 Results of the CBAs for a uniform nitrogen reduction, considering uniform or heterogeneous vulnerability
 394 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

395

396

397 This leads to an increase in the cost as well as in the benefits in order to arrive at a net
 398 benefit that is almost five times greater. The result is logical, as illustrated in Fig. E.2.

399 When the impact on the environment appears to be low, it may seem to be too costly to
 400 reduce fertilizer inputs compared to the damage avoided, thus generating a smaller

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

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

416

417 *4.2. CBAs for a spatialized reduction of nitrogen depending on the agricultural area*

418

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

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

439

440 **Table 3**

441 Results of the CBAs for a spatialized nitrogen reduction, considering uniform or heterogeneous vulnerability
 442 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
443 Net benefit (€/year)	127 505	642 373	124 028	609 606

444

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

447 will have to make greater efforts, whereas those of the Kolbsenbach and of the downstream
448 Souffel will have to make less and perhaps none at all. The additional effort on the
449 Leisbach can be explained by a high diffusion coefficient: 2.1% (result (2)), compared to
450 the other areas (the difference in the diffusion coefficient can be explained by the
451 pedology, the average distance between plots, etc.). It signifies that for 100 kg of nitrogen
452 applied in this area, 2.1 kg of nitrate are found in the water. Consequently, reducing the
453 nitrogen upstream will have a greater impact on the environment. For the area of the
454 Musau, the additional effort can be explained by a relatively weak flow (Fig. A.1). The
455 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
456 Kolsenbach, the Leisbach, the upstream Souffel and the downstream Souffel,
457 respectively. The dilution effect is therefore lower on the Musau, meaning that the same
458 mass of nitrate from the upstream agricultural areas will produce a higher concentration.
459 Inversely, reducing the nitrogen upstream will reduce the nitrate concentration even more
460 on this river. In contrast, the lesser effort can be explained by a very low diffusion
461 coefficient on the Kolbsenbach: 0.7%, and a very high average flow on the downstream
462 Souffel (Fig. 2). These “transformation/transport” effects within the catchments and the
463 “dilution” in the reaches from upstream to downstream therefore explain, regardless of the
464 damage scenario, that the effort will be greater on the Leisbach River, followed by the
465 Musau River, and then the upstream Souffel River. Compared to a uniform reduction effort
466 (Table 2), spatializing reduction efforts does not significantly increase the differences
467 between the results of the CBAs obtained for uniform or heterogeneous vulnerability
468 scenarios (Table 3). The effort between sub-catchments is slightly more balanced for the
469 heterogeneous vulnerability scenario. This can be explained by the fact that vulnerability
470 linked to weaker uses downstream due to a stronger dilution capacity is compensated for

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

473

474 *4.3. Advantage of spatializing the fertilization reduction effort*

475

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

483

484 **Table 4**

485 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%

486

487

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

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

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

499

500

501 **5. Discussion**

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

505

506 *5.1. CBAs for the optimal reduction of non-point source pollution*

507

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

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

520 Concerning nitrates, the WFD defines a standard of 50 mg/l for water bodies if they are to
521 have good status in terms of this parameter. In France, this standard is reached when the
522 50-mg/l concentration is not exceeded for more than 10% of the measurements. Thus, in
523 Table 5 below, we are interested in the number and percentage of reaches on the Souffel
524 where the optimal solution generates values that exceed this 50 mg/l standard for our eight
525 scenarios.

526

527 **Table 5**

528 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%

529

530
531 We can observe that good status is achieved, or almost, for the high range of the damage
532 estimate (3M), but only for 1/3 of the reaches with a nitrate concentration above 50 mg/l
533 for the low range (1M). This means that the WFD exemption would be allowed for a low
534 damage estimate, with pollution that could remain at the current level, i.e., 74 of the
535 reaches with concentrations above 50 mg/l, or 2/3 of the reaches (Fig. 2), whereas the
536 abatement effort should be made with the high estimate.

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

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

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

557

558 *5.2. Advantage of spatializing the abatement effort*

559

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

564 the receptors) and the vulnerability of the receptor sites (magnitude of the damage
565 generated according to the concentration) when the policy aims at maximizing the net
566 environmental benefit of the abatement costs (first-best), and of the transfer coefficients
567 and the intensity of the quality standard when the policy aims at achieving an ambient
568 standard at the minimum abatement cost (second-best). However, the benefit of
569 spatialization is context-dependent. Indeed, Tietenberg (2006) presents the results of
570 empirical studies on air pollution reduction policies in the United States where it appears
571 that the gains in pollution control costs to reach the same standard are reduced, depending
572 on the study, by 6.6% to 95.7% with a spatialized vs. uniform policy. Several authors have
573 studied the contextual parameters that could influence the benefit of spatialization (O’Ryan,
574 2006; Destandau and Nafi, 2010; Krysiak and Schweitzer, 2010; Destandau, 2013).

575 Our paper contributes to this literature by measuring the benefit of spatializing the
576 abatement effort under different assumptions about the damage generated by nitrate
577 pollution, including the magnitude of the damage at a low range (1M) and a high range
578 (3M). This makes it possible to study the benefits of spatialization according to the policy
579 issue.

580 In a theoretical study, Destandau and Nafi (2010) show that the advantage of spatialization
581 decreases when the standard is more restrictive in the case where the marginal costs of
582 cleanup are convex. In fact, for very stringent standards, the pollution control costs are so
583 high that the relative difference between uniform and spatialized regulation becomes
584 negligible. On the other hand, for marginal linear abatement costs, the standard has no
585 impact on the benefit of spatialization. In another theoretical study, Destandau (2013) once
586 again attempted to identify the impact of the environmental policy issue on the advantage
587 of spatialization, but this time in a first-best regulation where one seeks to maximize the
588 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.

636

637

638

639 **6. Conclusion**

640

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

653 The magnitude of damage and, consequently, the perception of negative effects generated
654 by the presence of nitrate, has a very big impact on the intensity of the reduction of N
655 fertilizers necessary to reach optimal concentrations. Actually, for a damage multiplied by
656 3, the optimal abatement effort is multiplied by 6, and the abatement costs by 14.
657 Moreover, the optimal average nitrate concentration in the basin decreases from 47 mg/l to
658 42 mg/l, which would allow the good status of the WFD to be achieved without an
659 exemption. In addition, the shift from the low to the high damage range also increases the
660 benefit of spatializing the abatement effort by a ratio of 1 to 9 for the absolute benefit, and
661 from 1 to 2 for the relative benefit. In the case of spatialization, an additional abatement
662 effort will be required for farmers whose effluents have a higher diffusion coefficient or
663 whose effluents drain into reaches of the river with weaker flows. Our results also show

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

677 This study mainly focused on scenarios related to damage and the spatialization of the
678 fertilizer reduction effort. Future studies could perhaps target different scenarios linked to
679 the cost of reducing nitrogen fertilizers. For example, rather than simply reducing the
680 quantity of nitrogen fertilizers, a change in agricultural practices could be considered. Crop
681 rotations that include legumes, in particular, could lead to a significant reduction in the use
682 of fertilizers. These rotations could also make it possible to decrease the use of pesticides
683 that impact the same water bodies, and could generate other benefits in terms of
684 biodiversity, for example. Likewise, we can ask ourselves if the cost of policies aimed at
685 modifying behaviors and practices should be included in decontamination costs as well.

686 These reflections would also make it possible to examine the opportunity of spatializing
687 the abatement effort as a function of the magnitude and heterogeneity of the cleanup and
688 policy costs. Finally, the approach proposed can be further enriched if the required data are

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

692

693

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809

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 γ_j : Number of nitrate concentration units above 25 mg/l

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

830 π_i : Diffusion coefficient = share of N_i found in the form of nitrate in the rivers

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

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

833

834 **Appendices**

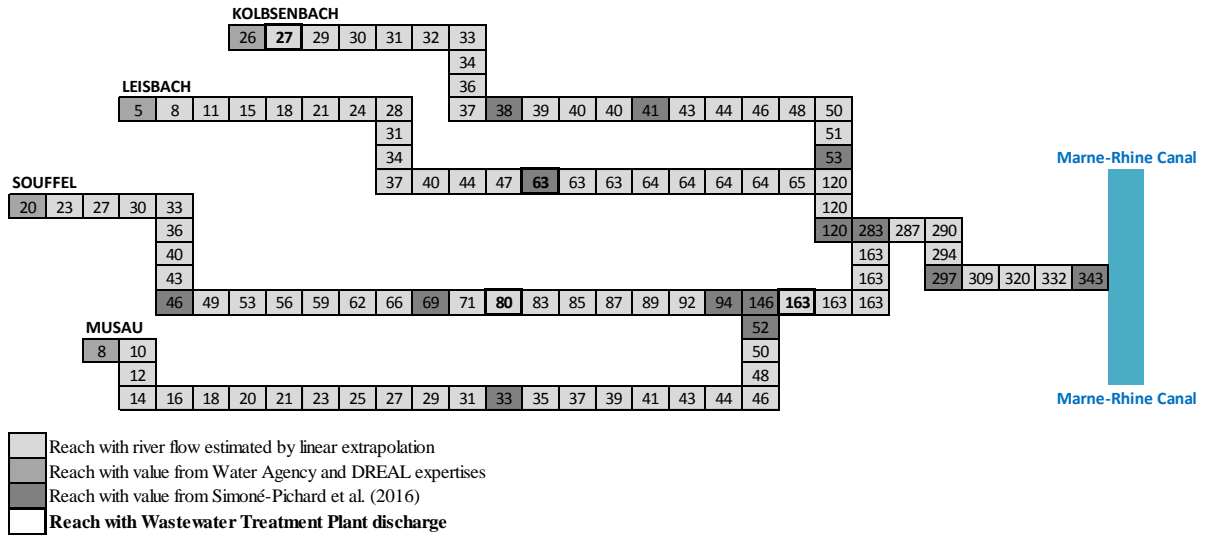
835

836 **Appendix A**

837

838 **Figure A.1**

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



840

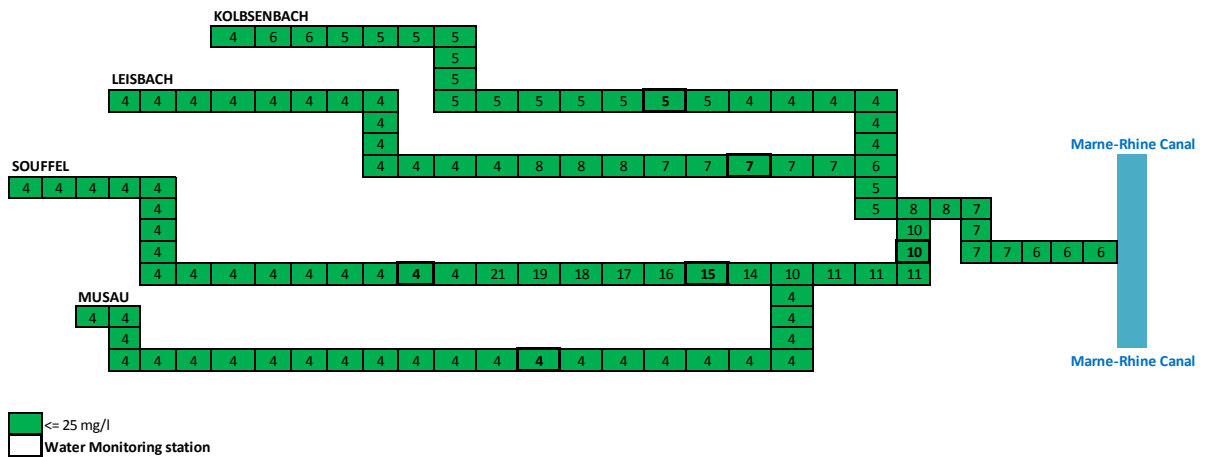
841 **Legend:** Our estimation of the average flow for each reach was based on several sources: Simoné-Pichard et
 842 al. (2016) who estimated the average flow for 14 catchment reaches; expertises of the “DREAL Grand Est”
 843 (Regional Direction of the Environment, Development and Housing for the Grand-Est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

848

849

850 **Figure A.2**

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



852

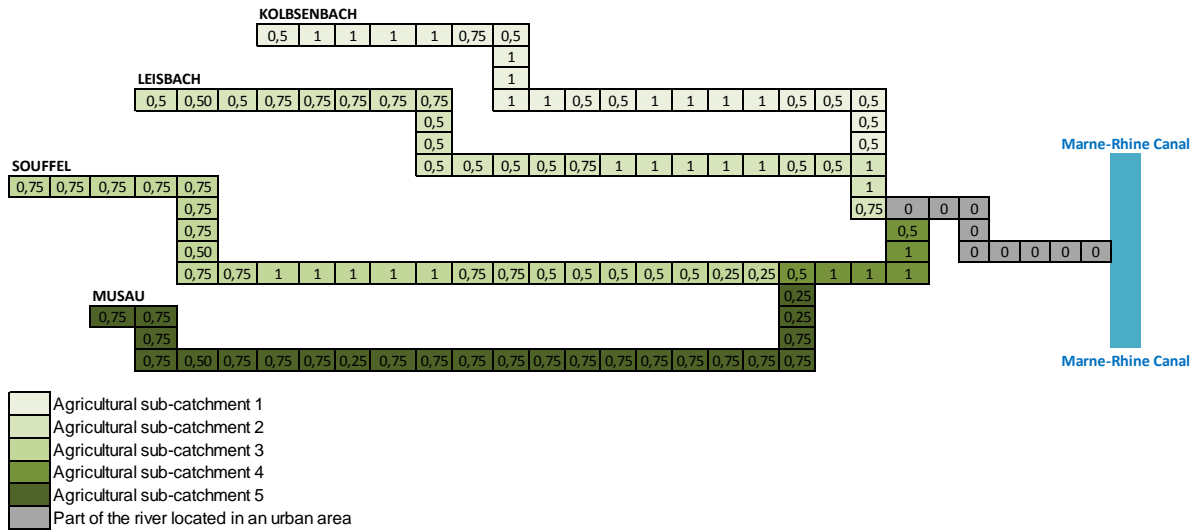
853

854 Appendix B

855
856

857 **Figure B.1**

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



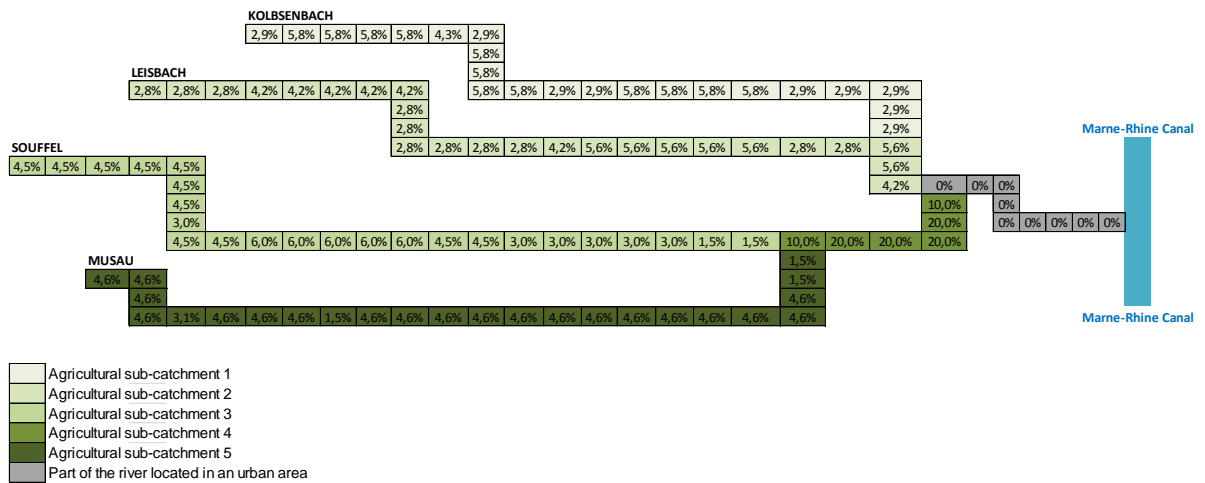
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861

862 **Figure B.2**

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



864

865

866 Appendix C

867

868 **Table C.1**

869 Construction of the cost of reducing farm N inputs.

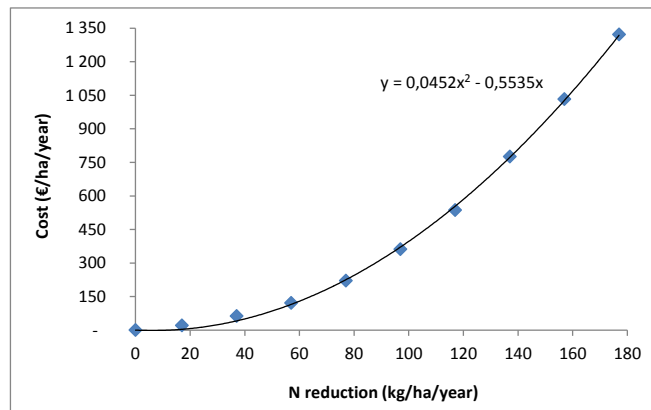
N application (kg/ha)	Corn Production (kg/ha)	Corn Sales (€/ha)	Cost N (€/ha)	Benefit (€/ha)	N reduction (kg/ha)	Cost of reduction (€/ha)
0	2 400	396	-	396	177	1 322
20	4 300	710	24	686	157	1 033
40	6 000	990	48	942	137	776
60	7 600	1 254	72	1 182	117	536
80	8 800	1 452	96	1 356	97	362
100	9 800	1 617	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	-

870

871

872 **Figure C.1**

873 Cost according to the reduction of N inputs.



874

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

880 Using polynomial regression, we obtain for each hectare:

881
$$“Cost\ of\ reduction” = 0.0452 \times (“N\ reduction”)^2 + 0.5535 \times (“N\ reduction”)$$

882 Appendix D

883

884 **Table D.1**

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

Author	Context	Low range (€/year)	Single estimate (€/year)	High range (€/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.

886

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

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

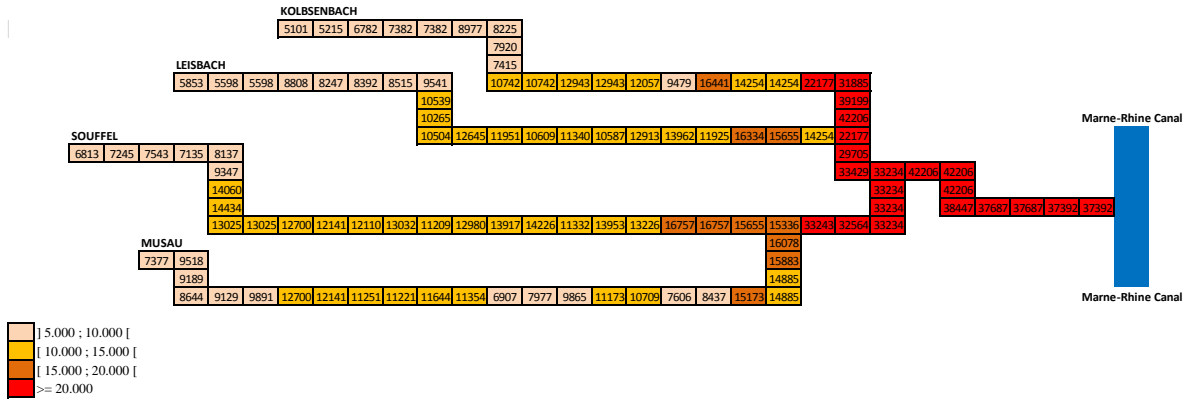
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899 Appendix E

900

901 **Figure E.1**

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



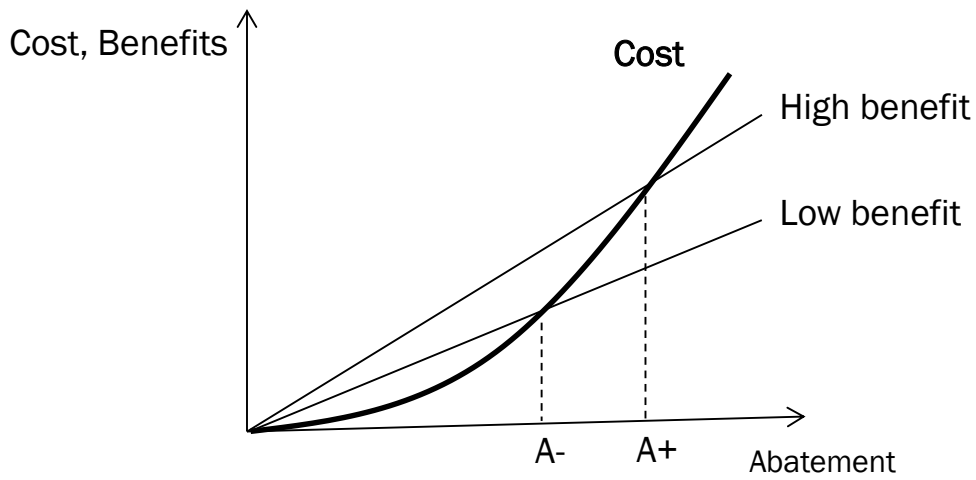
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906 **Figure E.2**

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



908

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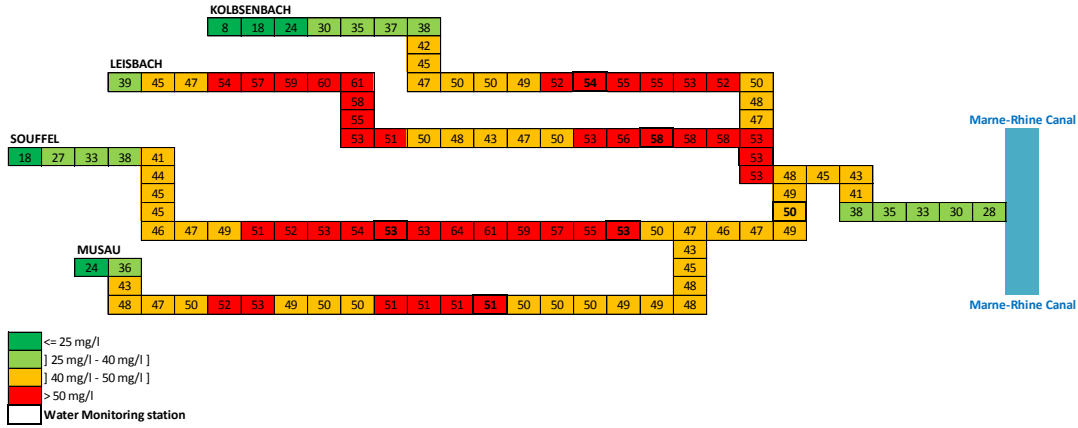
910 Appendix F

911 Optimal nitrate concentrations on the Souffel for each reach.

912

913 **Figure F.1**

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

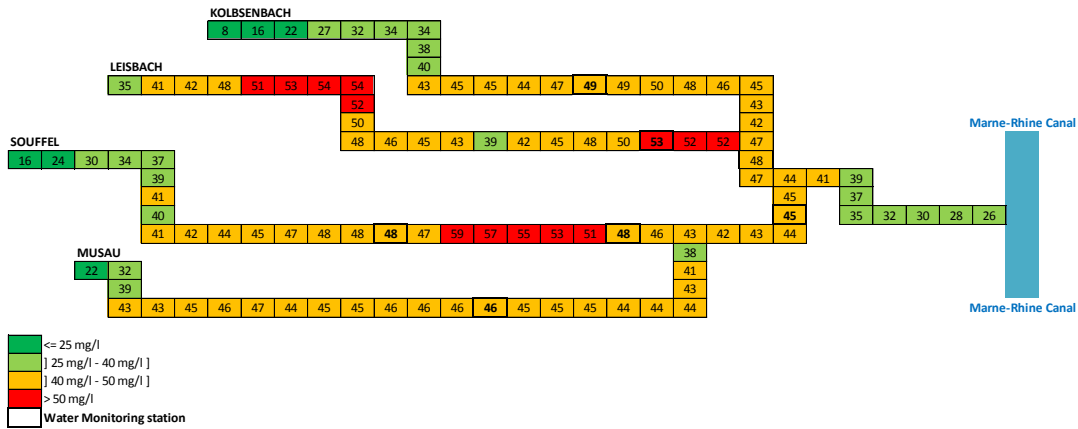


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917 **Figure F.2**

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



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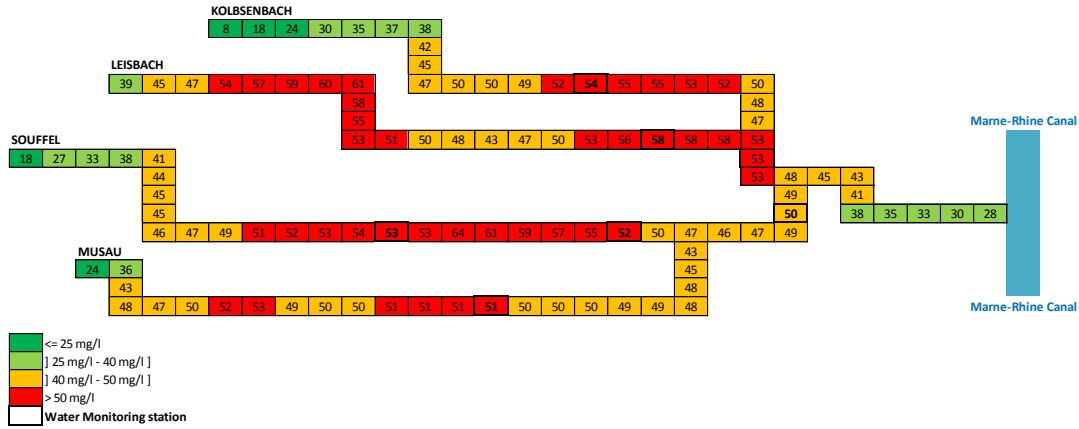
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Figure F.3

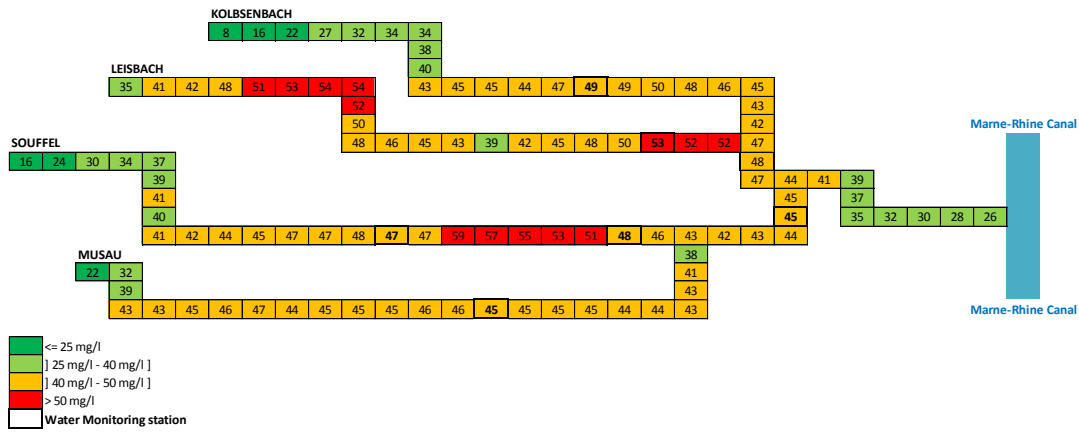
Reduction of the UNIFORM effort, HETEROGENEOUS vulnerability and total damage for the 1M scenario.



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Figure F.4

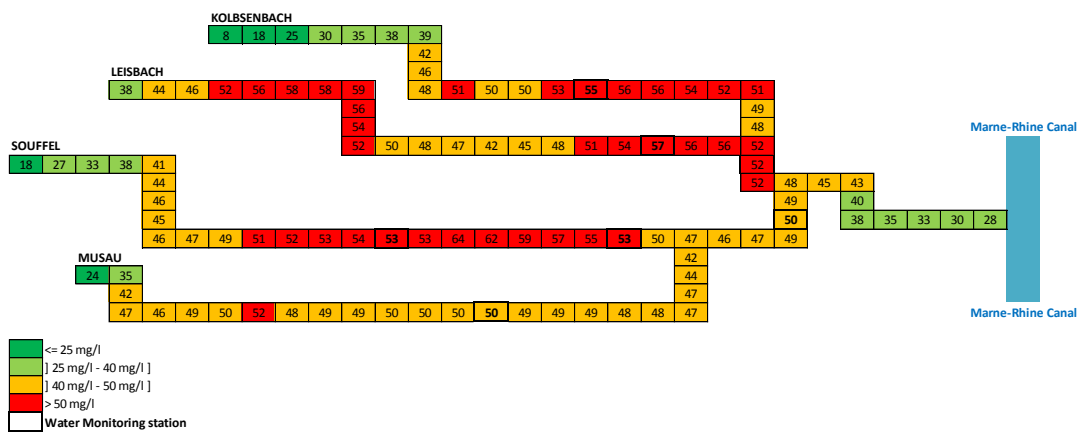
Reduction of the UNIFORM effort, HETEROGENEOUS vulnerability and total damage for the 3M scenario.



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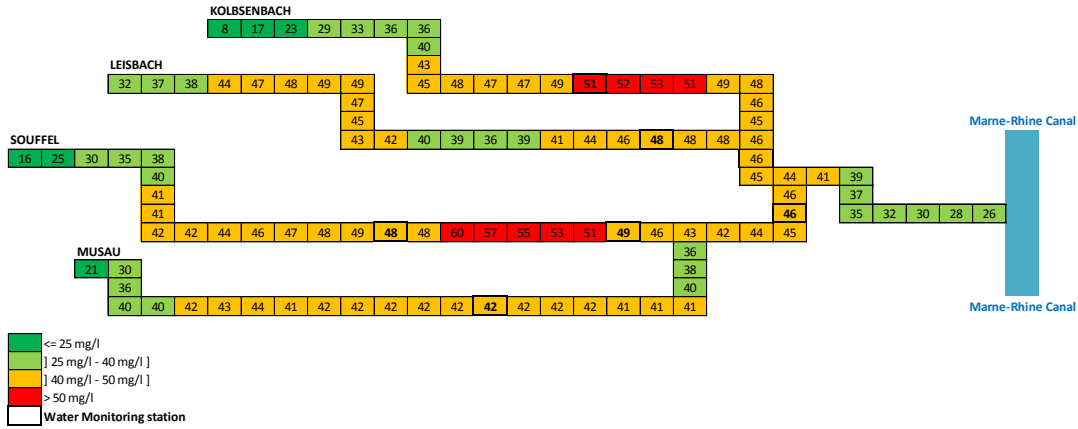
Figure F.5

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

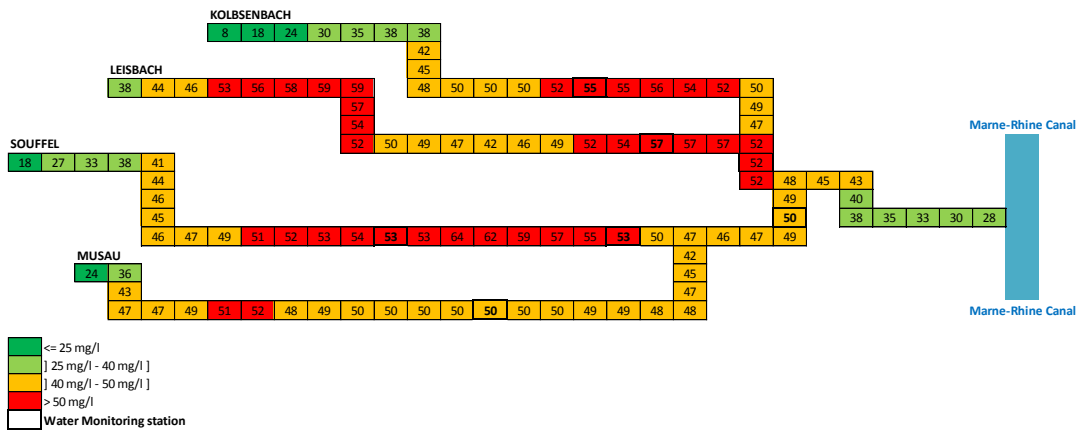


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940

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



943
 944
 945 **Figure F.7**
 946 Reduction of the SPATIALIZED effort, HETEROGENEOUS vulnerability and total damage for the 1M
 947 scenario.



948
 949
 950 **Figure F.8**
 951 Reduction of the SPATIALIZED effort, HETEROGENEOUS vulnerability and total damage for the 3M
 952 scenario.

