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Elucidating heterogeneous nitrate contamination in a small basement aquifer. A multidisciplinary approach: NO₃ isotopes, CFCs-SF₆, microbiological activity, geophysics and hydrogeology

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► **To cite this version:**

Emmanuelle Petelet-Giraud, Nicole Baran, V. Vergnaud-Ayraud, Angélie Portal, C. Michel, et al.. Elucidating heterogeneous nitrate contamination in a small basement aquifer. A multidisciplinary approach: NO₃ isotopes, CFCs-SF₆, microbiological activity, geophysics and hydrogeology. Journal of Contaminant Hydrology, 2021, 241, pp.103813. 10.1016/j.jconhyd.2021.103813 . hal-03336986

HAL Id: hal-03336986

<https://hal.science/hal-03336986>

Submitted on 9 May 2023

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1 **Elucidating heterogeneous nitrate contamination in a small**
2 **basement aquifer. A multidisciplinary approach: NO₃ isotopes,**
3 **CFCs-SF₆, microbiological activity, geophysics and**
4 **hydrogeology.**
5

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17 **Highlights**

- 18 • Heterogeneous nitrate groundwater contamination in a small basement-rock aquifer
19 • Nitrate contamination is controlled by anthropogenic activities and hydrogeology
20 • Past land use has a significant remanent effect on nitrate concentration in
21 groundwater
22 • Dual isotope approach of NO₃ revealed its source and denitrification processes
23 • Multidisciplinary approach allows deciphering driving mechanisms of contamination
24

25
26 **Keywords**

27 Nitrate contamination, basement aquifer, nitrate isotopes, groundwater residence time,
28 microbiological activity, Electrical Resistivity Tomography geophysics, Plourhan catchment
29
30

31 **Abstract**

32 Nitrate contamination of groundwater remains a major concern despite all the measures and
33 efforts undertaken over the last decades to protect water resources. We focused on a small
34 catchment in Brittany (France) facing nitrate pollution with concentrations over the European
35 drinking water standard of 50 mg.L⁻¹. This is a common situation in catchments where -
36 supposedly effective - measures were applied for reducing the transfer of N to groundwater.
37 At the scale of this small (~100 ha) basement aquifer, nitrate concentrations are very
38 heterogeneous in the groundwater, sampled up to 15-20 m below the soil surface in several
39 observation wells (hereafter referred as piezometers) and up to 110 m deep in a borehole
40 drilled through a faulted area near the Spring (outlet of the catchment). We used

41 complementary and robust approaches for exploring and better to better constraining the
42 driving parameters of nitrate transfer and distribution in groundwater. Detailed geological
43 work and a geophysical electrical resistivity tomography survey identified the lithologies,
44 tectonic structures and weathering layers. This highlighted a complex geological structure
45 with several compartments delimited by faults, as well as the highly variable thickness of the
46 weathered layer. It also illustrated the heterogeneity of the hydrosystem, some
47 compartments appearing to be disconnected from the general groundwater flow. This was
48 confirmed by geochemical analyses and by the mean apparent groundwater residence time
49 based on CFCs-SF6 and noble-gas analyses, locally revealing old and nitrate-free
50 groundwater, and very old water with a recharge temperature below than the current average
51 temperature in the area, reflecting water dating back to the last period of glaciation (-19 to -
52 17 ky). Nitrate isotopes clearly showed denitrification processes in a few piezometers, which
53 was generally supported by microbiology and molecular biology results. This highlighted the
54 presence of functional genes involved in denitrification as well as a capacity of the
55 groundwater microbial community to denitrify when *in situ* conditions are favourable. This
56 type of combined approach - covering chemistry, isotopic methods, dissolved gases,
57 microbiological activity, geophysics and hydrogeology - appears to be indispensable for
58 implementing the most relevant programme of measures and for accurately assessing their
59 effectiveness, notably by considering the timeframe between implementation of the
60 measures and their impact on groundwater quality.

61 **1 Introduction**

62 Nitrate contamination of groundwater remains a major concern despite all the efforts to
63 protect waters aquifers against nitrate pollution, such as implementation of the European
64 Council Directive [91/676/EEC](#) adopted in 1991, and of the European Water Framework
65 Directive [2000/60/EC](#) and its daughter Directive [2006/118/EC](#). In addition to agronomic
66 measures, a key issue is to arrive at better understanding the driving parameters of nitrate
67 transfer towards, and distribution in, groundwater, i.e. the geological structures and the
68 hydrogeological functioning.

69 The correlation between an increased use of (synthetic and organic) fertilizers and the
70 increase of nitrate concentrations in groundwater has been known for a long time (e.g.
71 [Aquilina et al., 2012](#)) but uncertainties remain concerning the transit time between topsoil
72 and aquifer. In particular, the immobilization/remobilization processes of N governing
73 nitrogen cycling and nitrate transfer through soil are still poorly constrained in terms of “delay
74 effect”. [Sebilo et al. \(2013\)](#), who investigated the fate of a single application of an isotopically
75 labelled nitrogen fertilizers for three decades in an experimental field, concluded that
76 attempts for reducing agricultural nitrate contamination of aquatic systems must consider the

77 long-term legacy of past applications of synthetic fertilizer and the nitrogen-retention capacity
78 of agricultural soils.

79 In addition to nitrogen-retention and -cycling processes in soil, the N stock – mainly nitrate –
80 in the unsaturated aquifer zone must be considered (Baran et al., 2007). Nitrate being highly
81 soluble, it is generally considered that it will transfer through the unsaturated zone towards
82 groundwater at the same rate as infiltrating water. This implies that a good understanding, of
83 the subsurface geology and its geometry as well as of the hydrogeological functioning of the
84 aquifer system is essential, as they - at least partly - govern pollutant transfer time at the
85 catchment scale. A proper understanding of the hydrogeological functioning can be obtained
86 from groundwater residence-time assessment, using chlorofluorocarbons (CFCs) and
87 sulphur hexafluoride (SF₆) gases for studying modern (< 60 yr) groundwater dynamics and
88 mixing processes. These anthropogenic gases have been released into the atmosphere
89 since the 1950s and are now found in natural waters, with atmospheric equilibria that allow
90 groundwater dating based on their atmospheric concentrations chronicles (e.g. Busenberg
91 and Plummer, 1992; Ayraud et al., 2008; Aeschbach-Hertig and Solomon; 2013; Chatton et
92 al., 2016).

93 Furthermore, part of the N can leave the nitrogen cycle as gases (N₂O and N₂) resulting from
94 denitrification, governed by heterotrophic or autotrophic denitrifying bacteria transforming
95 NO₃ to N₂ (Stein and Klotz, 2016). This process affects the global N budget, especially the
96 nitrate one, and must be considered as a major process when evaluating of the fate of N
97 from soil to groundwater. Nitrogen and oxygen isotopes of nitrate ($\delta^{15}\text{N-NO}_3$ and $\delta^{18}\text{O-NO}_3$)
98 are frequently used as tracers of nitrate sources in groundwater, as the various
99 synthetic/organic N fertilizers have different isotopic compositions (Kendall et al., 2007).
100 However, nitrate isotopes also are ideal tools for tracing N transformation processes, such as
101 denitrification (Böttcher et al., 1990), as NO₃ reduction to N₂ is a highly fractionating process
102 (Kendall and Aravena, 2000) leading to a joint increase of $\delta^{15}\text{N-NO}_3$ and $\delta^{18}\text{O-NO}_3$, whereas
103 simple dilution, also leading to a decrease in NO₃ concentration, does not modify the isotopic
104 signatures.

105 We designed and implemented an innovative interdisciplinary study, coupling complementary
106 and robust approaches for defining the spatio-temporal nitrate distribution in a small
107 catchment of a basement-rock aquifer. For that purpose, we had to answer the following
108 questions. (1) How is groundwater quality—especially for nitrate—controlled spatially and
109 temporarily, by natural water-rock interactions and by anthropogenic input related to
110 agricultural practises and its evolution over the past decades (hydrogeological, geophysical
111 and chemical approaches)? (2) What are the nitrate sources and their potential attenuation,

112 as shown by N and O isotopes of NO₃ fingerprinting, and through assessing the
113 denitrification potential by monitoring nitrate-reducing bacteria genes (isotopic and
114 microbiological approaches)? (3) What is the groundwater residence-time distribution, in
115 order to evaluate the hydrodynamics and the potential spatial variability of the aquifer, and as
116 a supplementary constraint to support the hypothesis regarding the chemical and isotopic
117 compositions (CFC-SF₆ and rare dissolved gases approach)?

118

119 **2 Site description**

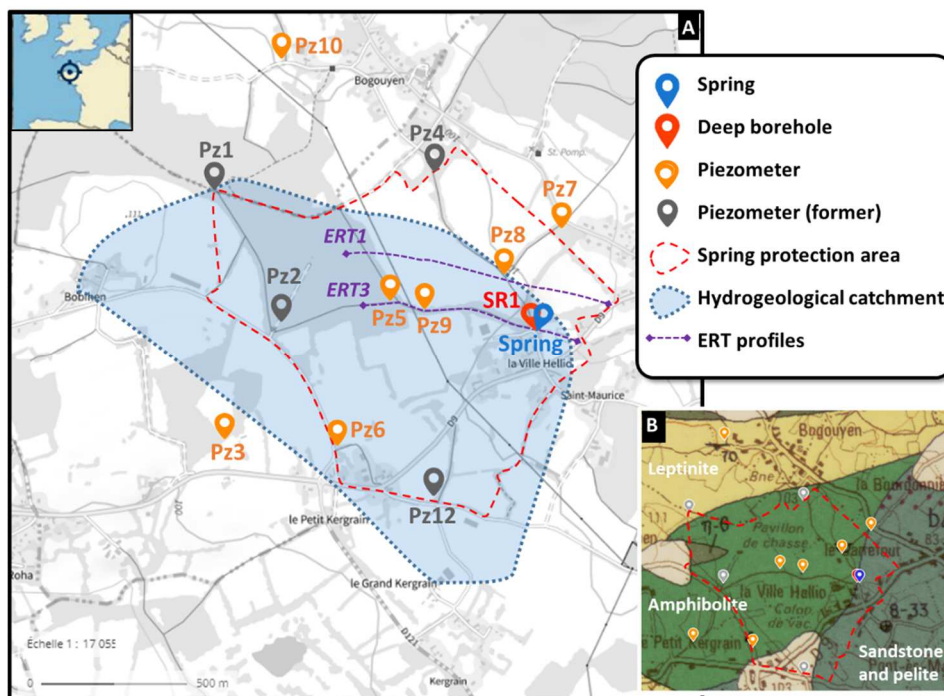
120 The Plourhan site we studied is a small (1.26 km²) hydrogeological catchment. In 2006, a
121 potentiometric map was drawn for resource-protection purposes, relying on different
122 hydrogeological approaches such as the drilling of 12 piezometers and a deep borehole. It is
123 located 10 km from the coast in Brittany (northwest France). Geologically, it is underlain by
124 Brioverian basement composed of metasedimentary pelite and sandstone, and amphibolite,
125 both overlain by an 8 to 20 m thick saprolite layer. The basement that can be highly
126 fractured, especially the amphibolite. A major fault forms the boundary between
127 metasediments and amphibolite, with a potentially wide fracture zone (Figure 1). This
128 description, based on the 1:50,000-scale geological map was not precise enough in view of
129 the small catchment size. For this reason, a detailed geological survey of the catchment was
130 part of this study as was a geophysical survey, providing a better definition of the detailed
131 geology and its geometry (see Supplementary material). In addition, a pedological study
132 carried out during delineation of the area to be protected, showed relative soil uniformity of
133 the site. The soil is brown and moderately deep (40 cm on average in the east of the site and
134 about 80 cm in the southwest). Under the forest, to the northwest, a particular pedogenesis is
135 observed, the soil being pseudo-gley.

136 A natural spring emerges near the major fault and has been exploited since 1968 via a 4.4 m
137 deep borehole that taps the water coming from the saprolite layer. This spring, hereafter
138 referred to as Spring, feeds a small stream that exits the basin to the east. In 2006, 12
139 piezometers (Pz1 to Pz12), between 10 and 22 m depth, were drilled mainly for delineating
140 the catchment. In these piezometers, the groundwater level is few metres below the soil level
141 (bsl). They are screened all along the weathered layer down to the top of the “fresh”
142 basement. The saturated zone is only a few metres thick. A few measurements before our
143 study showed strong heterogeneity of the groundwater quality. Monthly monitored water
144 levels between October 2006 and March 2008 were used for drawing potentiometric maps,
145 thus defining the hydrogeological contours of the catchment and identifying flow directions
146 (Fig. 1). Not surprisingly, the hydrological and topographic catchments have different
147 boundaries. Thus, for the piezometers accessible during this study, Pz3, 6 and 8 are located

148 upstream (respectively south and north of the Spring), while Pz5 and Pz9 are located in the
149 middle of the catchment. The others (Pz10 and Pz7) lie outside the catchment.

150 In addition to these 12 piezometers tapping water in the saprolite, a deep borehole (SR1),
151 drilled at about 30 m from the natural Spring within the fault zone, has a total depth of 133 m.
152 It is screened below 50 m (bsl) and intersected paragneiss, and metamorphic sandstone and
153 pelite.

154 The water supply for the 2 000 inhabitants is based on the water pumped from the Spring
155 (Figure 1) which must be diluted in order to fulfil the drinking-water quality standards for
156 nitrate. Indeed, the Spring has nitrate concentrations exceeding the 50 mg/L drinking water
157 standard and the dilution takes place at the treatment plant with water from another borehole.

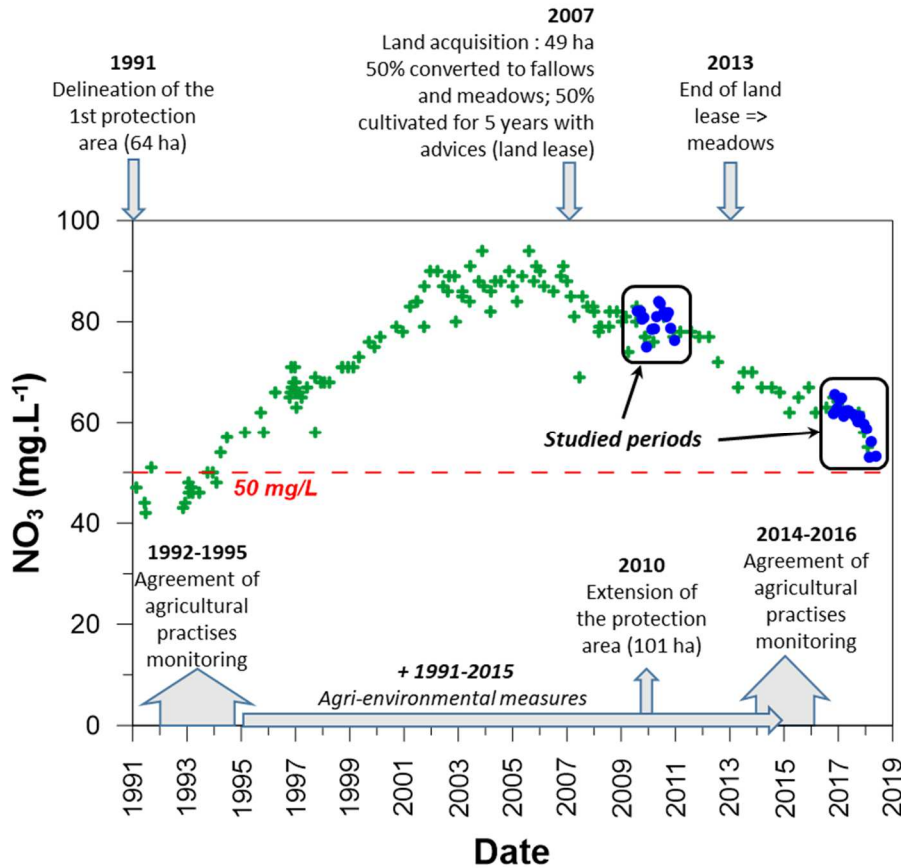


158
159 *Figure 1: (A) Location of sampling sites and geophysical investigations in the Plourhan site (NW*
160 *France; background 1:25,000-scale IGN map). Piezometers with grey symbols are no longer*
161 *accessible. (B) Geological map of the studied area (@BRGM Pontrieux sheet, 1:50,000 scale).*

162 A protection area for the groundwater catchment was administratively defined in 2010 along
163 parcel boundaries. It specified the regulation of (agricultural) land use, maintaining plots in
164 forest or meadow areas, and plots where annual crops were allowed with specific agronomic
165 recommendations. As of 2013, annual crops were no longer allowed and were replaced by
166 meadows (Figure 2).

167 Historically, the studied site has been subject to strong agricultural pressure. From 1980 to
168 1990, vegetable production dominated. In the mid-1990s, land use was mainly dedicated to
169 market gardening, with a significant surface for peas, potatoes and cauliflower. From the

170 mid-1990s to the present, in an effort to improve groundwater quality, farmers significantly
 171 reduced the vegetable areas (Figure 2), compensated by an increase in wheat, maize and
 172 especially grass. Despite these efforts, nitrate concentrations in Spring water only decreased
 173 slowly and still remain above the 50 mg/L standard (Figure 2).



174

175 *Figure 2: Temporal evolution of the nitrate concentrations in the Spring since the early 1990s (green crosses).*
 176 *Both studied periods (2009-2011 and 2016-2018) are shown as blue dots. Historical data are from the French*
 177 *national groundwater database (ades: <https://ades.eaufrance.fr>). History of the measures for improving water*
 178 *quality is indicated.*

179

180 In the area, historically and until recently, nitrogen was applied in both mineral and organic
 181 form, depending on the cultivated plot. Detailed information is not available, but it is known
 182 that in 2014, in the protection area, four of the seven farms produced animal excrements
 183 (cattle, pigs, poultry) and five had plots on which animal excrements, green-waste compost
 184 and/or household waste were spread (Côtes d'Armor Chamber of Agriculture, 2014).

185 A detailed knowledge of land use allows addressing the issue of why groundwater quality is
 186 spatially heterogeneous. Such groundwater catchment is representative of lithology
 187 (basement), soil and climate of France and Western Europe.

188

189 3 Material and methods

190

191 3.1 Sampling strategy

192 The study site was monitored for two distinct periods. The Spring was sampled on a monthly
193 basis from August 2009 to June 2010 (10 campaigns = 1st period) and from October 2016 to
194 July 2019 (28 campaigns = 2nd period), whereas piezometers were only sampled twice and
195 four times during the 1st and 2nd periods, respectively. It should be noted that some
196 piezometers were no longer accessible during the 2nd period. Prior to sampling, several field
197 parameters were measured, including electrical conductivity, pH, redox and oxygen content
198 (Table S1 Supplementary material). Major ions were analysed for all samples.
199 Microbiological parameters were measured in each samples from the 2nd period, as was
200 groundwater dating. Isotopic investigations only concerned samples from April 2017.

201

202 3.2 Chemical and isotopic investigations

203 3.2.1 Analytical methods

204 For both sampling periods, the groundwater samples were collected in polyethylene bottles
205 and filtered through 0.45 µm PVDF filters for chemical and nitrate isotopes analysis. Bottles
206 dedicated to cation analysis were acidified with 15N ultrapure HNO₃ to pH<2.
207 Physicochemical parameters of each sample were measured on site, including the electrical
208 conductivity (EC), standardized to 25 °C, water temperature, pH, dissolved oxygen (O₂ diss.)
209 and redox potential (Eh). Samples were stored at 4 °C in the dark prior to analysis; this took
210 place in the BRGM Laboratories by ICP-AES (Ca, Na, K, Mg; uncertainty <10%), ion
211 chromatography (Cl, SO₄, NO₃, uncertainty <10%) and titration method according to N EN
212 ISO 9963-1(HCO₃⁻; CO₃²⁻, uncertainty <5%).

213 Both δ¹⁵N and δ¹⁸O of NO₃ were measured by the AgNO₃ method at BRGM according to the
214 following procedure: The filtered water is chemically purified through cationic and anionic ion-
215 exchange columns. Nitrate is recovered as HNO₃ by HCl elution. The eluate is converted to
216 AgNO₃ by addition of silver oxide (Ag₂O): HCl + HNO₃ + Ag₂O => AgCl_(s) + AgNO₃ + H₂O.
217 The filtrate containing the nitrate, once freed from the other oxygenated compounds by
218 adding BaCl₂, is lyophilized to obtain solid AgNO₃ (Chang et al., 1999, Silva et al., 2000). The
219 solid AgNO₃ precipitate is then weighed into silver capsules that are inserted separately into
220 a furnace composed of a glassy carbon reactor heated to 1450 °C. The N₂ and CO gases
221 resulting from high-temperature combustion are driven by a helium flow to a chromatographic
222 column. This separates the gas peaks before their introduction into the source of the mass
223 spectrometer for determining the nitrate δ¹⁸O and δ¹⁵N isotopic ratios. The analytical
224 accuracy is ± 0.5 ‰ for δ¹⁵N and δ¹⁸O.

225 Water samples for chlorofluorocarbons and sulphur hexafluoride (CFCs and SF₆) analyses
226 were collected in stainless-steel ampoules (40 mL and 300 mL, respectively) after rinsing
227 them with at least three times the sample volume, without any contact with atmospheric air
228 during sampling. The CFCs and SF₆ concentrations, used for calculating of groundwater
229 residence times, were measured using a gas chromatograph with an electron capture
230 detector (GC-ECD) (Labasque et al., 2006; Ayraud et al., 2008), at the Plateforme
231 CONDATE Eau (OSUR, University of Rennes 1, France) for the 2nd period. The
232 concentrations of CFC-11, CFC-12 and CFC-113 were measured with an analytical
233 uncertainty of 1%, and the SF₆ with an uncertainty of 5% (Labasque et al., 2006; Ayraud et
234 al., 2008). Gas concentrations were used for calculating atmospheric mixing ratios (pptv),
235 and compared to the atmospheric evolution curve (NOAA HATS program), to determine the
236 apparent groundwater piston age. For each sampling point, three hypothetical lump
237 parametric models, a piston-flow model (PFM), an exponential model (EM), and a binary
238 mixing model (BMM), were systematically tested to determine the apparent groundwater
239 residence time (Matoszewski and Zuber, 1996). The overall uncertainty of derived
240 groundwater-age estimates, including sampling biases and analytical errors, as well as the
241 errors resulting from uncertainty of the recharge temperature and considering dispersion-
242 adsorption effects, is estimated to be ±3 years (Labasque et al., 2014). For the first sampling
243 period (2009-2011) a simplified sample collection method was used (IAEA, 2006). The
244 analyses were made by Spurenstofflabor at Wachenheim (Germany), using gas
245 chromatography with an electron-capture detector after pre-concentration using the purge-
246 and-trap technique. The detection limit was 10⁻⁴ pmol, allowing the measurement of CFC
247 concentrations down to 0.01 pmol.L⁻¹ and of SF₆ concentrations to 0.1 fmol. L⁻¹;
248 reproducibility was about ± 5% for water samples.

249 Major and noble gases were measured throughout the sampling campaign using a GC-TCD
250 (CG 3000, SRA instrument) after headspace extraction with a He gas phase following the
251 Sugisaki and Taki method (1987). The analytical error for Ne and Ar/N₂ measurements was
252 3% and 2%, respectively. The analysis of Ne/Ar ratios allows calculating excess air and
253 recharge temperatures (Heaton and Vogel, 1981).

254 3.2.2 N and O isotopes of nitrate theoretical background

255 Since the 2000s, nitrogen and oxygen isotopes of nitrates ($\delta^{15}\text{N-NO}_3$ and $\delta^{18}\text{O-NO}_3$) are used
256 for tracing nitrate sources in surface and groundwaters (Widory et al., 2005, 2013; Kendall et
257 al., 2007; Fenech et al., 2012; Puig et al., 2017). Nevertheless, N-sources discrimination is
258 often not straightforward as nitrate is not a conservative molecule, and thus the N
259 transformation process has to be considered for properly tracing N sources. N and O
260 isotopes are also valuable tracers for better constraining the N cycle from soil to

261 groundwater, i.e. immobilization, mineralization, nitrification and denitrification. These
262 different processes may thus lead to a loss of the original $\delta^{15}\text{N-NO}_3$ and $\delta^{18}\text{O-NO}_3$ isotopic
263 signatures through isotopic fractionation (Zhang et al., 2019). However, such isotopic
264 fractionations is an excellent tool for identifying these processes and thus for better
265 constraining N mobility.

266 Denitrification by oxidation of pyrite (FeS_2), i.e. during chemoautotrophic respiration of
267 sulphur-oxidizing *Thiobacillus denitrificans*, has been identified as an important denitrification
268 pathway in basement aquifers where pyrite is abundant (Pauwels et al., 2000). Note that
269 Aquilina et al. (2018) suggested another type of autotrophic denitrification: biotite dissolution
270 with the oxidation of Fe(II) contained in biotite. Denitrification is a highly fractionating process
271 (Böttcher et al., 1990; Kendall and Aravena, 2000), following a Rayleigh distillation model,
272 leading to a joint increase of the isotope ratios $\delta^{15}\text{N-NO}_3$ and $\delta^{18}\text{O-NO}_3$ of the residual NO_3 .
273 The N cycle in soil further modifies the isotopic signature of the nitrate leached out and
274 transported toward groundwater (Mengis et al., 2001, Sebiló et al., 2013). The newly formed
275 nitrate through ammonium nitrification is generally assumed to take nitrogen from NH_4^+
276 (and/or NO_2), while one oxygen comes from atmospheric O_2 and two from water (H_2O)
277 considering a simple stoichiometric relationship (Equation 1; Kumar et al., 1983; Kendall,
278 1998 and references therein); Note that Buchwald and Casciotti (2010) added constraints to
279 this simple model considering oxygen isotope exchanges of intermediate species with
280 ambient water, during nitrite oxidation corresponding to the final step of nitrification process.

281
$$\delta^{18}\text{O-NO}_3 = \frac{2}{3} (\delta^{18}\text{O-H}_2\text{O}) + \frac{1}{3} (\delta^{18}\text{O-O}_2) \quad (\text{Equation 1})$$

282

283 3.3 Microbiological and biomolecular approaches.

284 The spatial and temporal evolution of the abundance of bacteria involved in nitrate reduction
285 was assessed through measurements of nitrate reductase encoding genes *narG* and *napA*
286 copies. In November 2016, April and October 2017, and May 2018, water (6 L) was collected
287 from each piezometer and filtered (0.22 μm) to collect biomass. Microbial DNA was extracted
288 from the filters using a FastDNA™ Spin Kit for Soil (MP Biomedicals, USA) according to
289 manufacturer's recommendations, with a FastPrep®-24 at a speed of 5 m.s^{-1} during 30 s,
290 and quantified with a Quantifluor dsDNA sample kit and the Quantus fluorimeter, according
291 to the manufacturer's instructions (Promega, USA). The abundances of *narG* and *napA* gene
292 copies was determined by real-time quantitative PCR (qPCR) according to Bru et al. (2007).

293

294 In order to assess potential denitrification activity, batch experiments were run on waters
295 sampled in April 2017 in the Pz3, Pz6, Pz8, Pz9 and Pz10 piezometers. Briefly, duplicate

296 300 mL flasks were filled in with 150 mL of water under anaerobic conditions (nitrogen
297 atmosphere). Acetate ($C_2H_3NaO_2$) was added as a carbon source in each flask at a final
298 concentration of 100 mg/L. Considering nitrate concentration in each well, nitrate was added
299 to the flasks to reach a final NO_3^- concentration of 100 mg/L. Abiotic tests in the presence of
300 sodium azide (Cabrol *et al.*, 2017) were also performed. The flasks were incubated for 20
301 days in the dark at 25 °C under stirring (100 rpm). Every 4 to 5 days, 8 mL of water was
302 sampled for nitrate, nitrite and acetate quantification, and 5 mL of the gas phase was
303 sampled for N_2O analysis. Nitrate and acetate were quantified by ionic chromatography
304 (Dionex IC3000-SP-EG-DC system equipped with an AS50 autosampler and a
305 conductimetric detector) according to the NF EN ISO 10304-1 (2009) method. Nitrite was
306 analysed by colorimetry according to the NF ISO 15923-1 (2014) method. N_2O was analysed
307 by gas chromatography (Varian CP-3800 GC equipped with a gas injection valve and an
308 electron capture detector).
309 For statistical analysis, boxplots were calculated using R4.0.1 and RStudio (n=2). Data were
310 analysed by the non-parametric Kruskal-Wallis test ($p>0.05$).

311

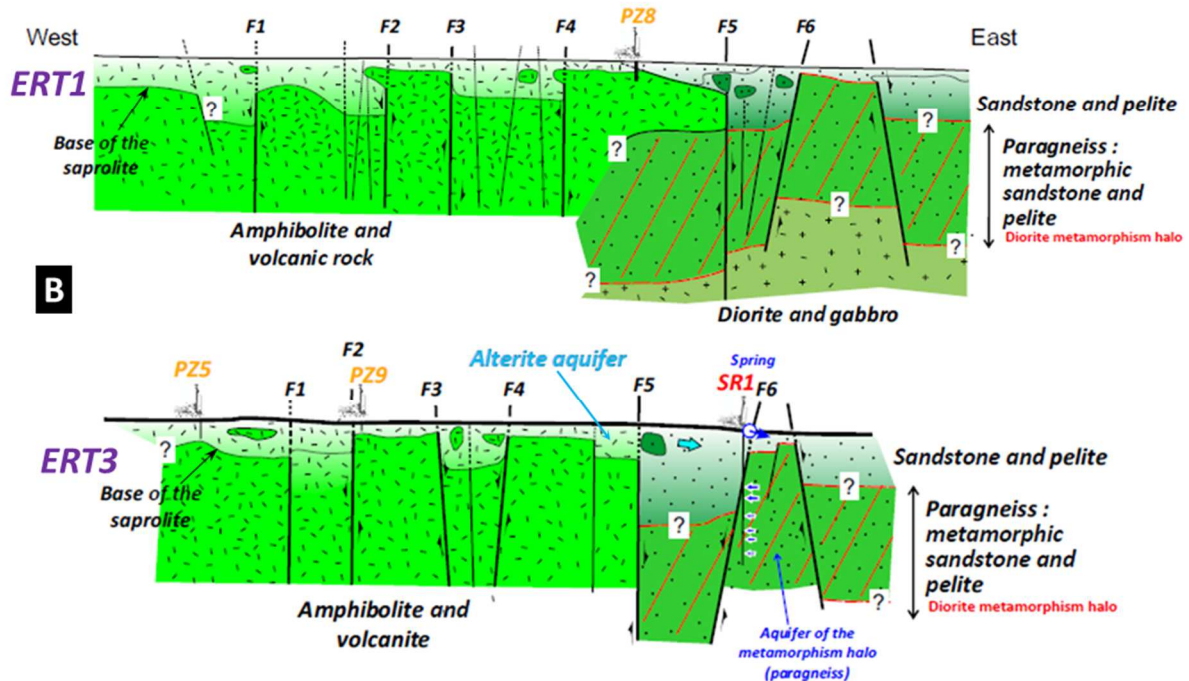
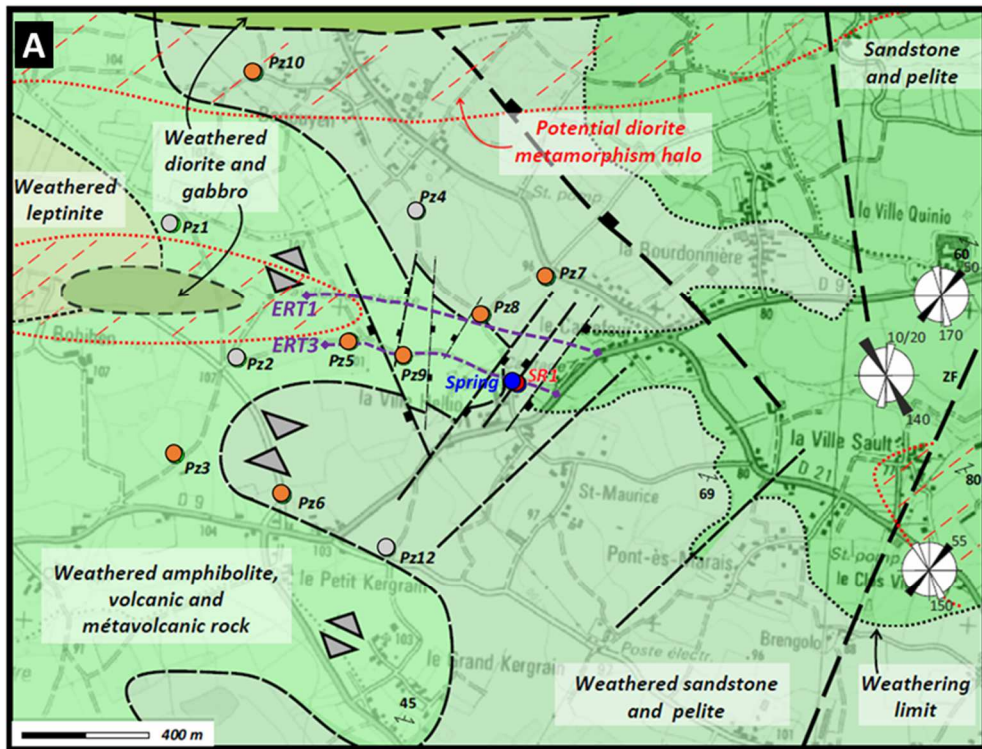
312 **4 Results and discussions**

313

314 **4.1 Improving of the hydrogeological model**

315 Geological investigations consisted in accurately identifying the rock types, tectonic
316 structures and degree of weathering, in order to refine the geological map at the scale of this
317 small catchment and to better understand the hydrogeological functioning. This field work
318 confirmed that the catchment is located on a Brioverian anticlinal axis oriented N100/110,
319 intersected by faults oriented N150, sub N-S and N050. The weathering profile forms a
320 paleosurface at an altitude of about 80 m above mean sealevel, and seems to be affected
321 by faulting.

322



323

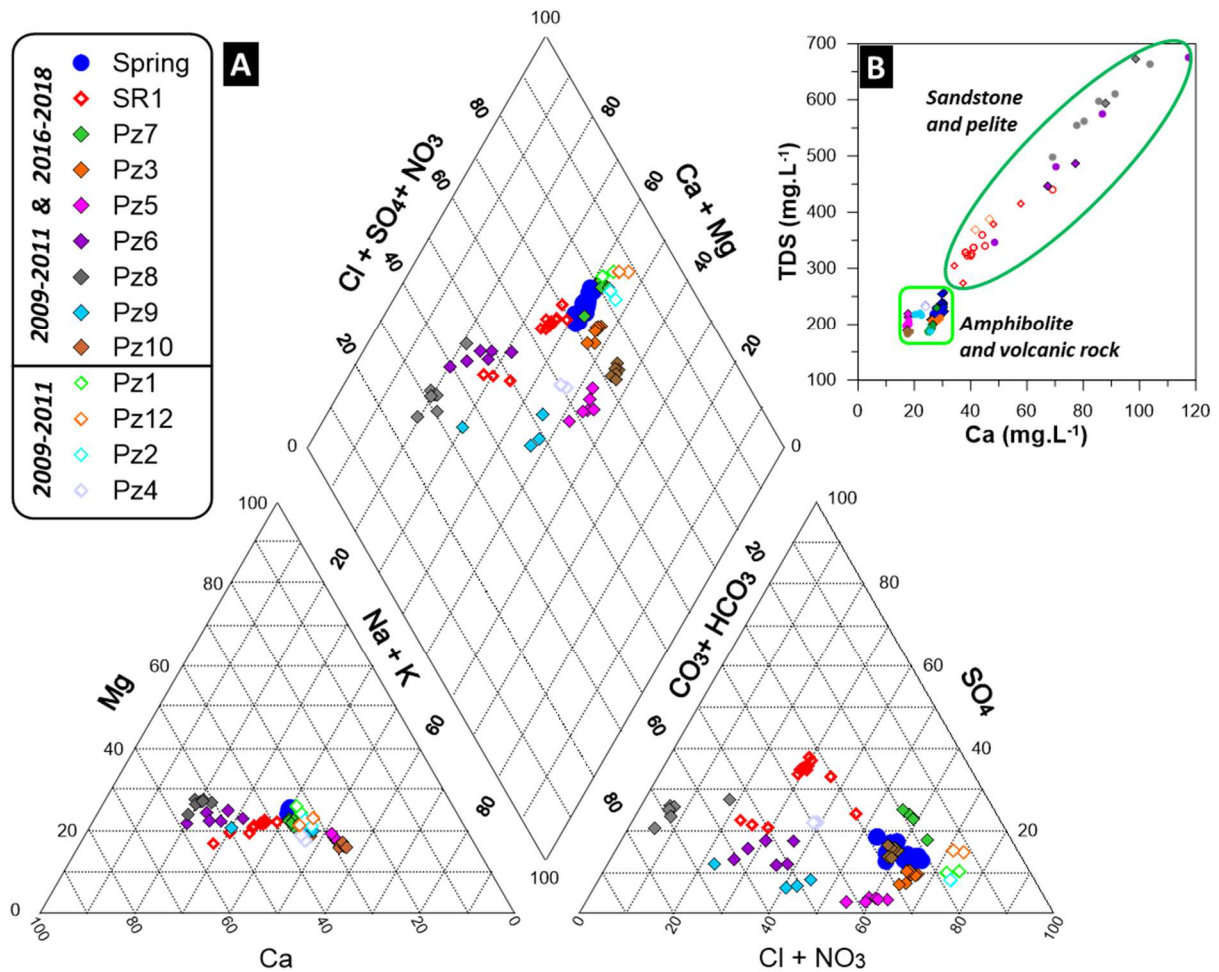
324 Figure 3: (A) Revised geological map of the study area. (B) Geological cross-sections derived from the electrical
 325 resistivity distribution profiles (FIG S1 in supplementary material).

326 Geophysical results showed major horizontal resistivity contrasts, with two distinct domains
 327 separated by a significant vertical discontinuity (F5; FIG. S1 Supplementary material). The
 328 western resistive domain corresponds to amphibolite, while the conductive eastern unit
 329 corresponds to metamorphic sandstone and pelite (Egal et al., 1996; Figures 1 and 3A). To
 330 the west, a vertical resistivity gradient is interpreted as a signature of the amphibolite
 331 weathering profile consisting of alterite (C0, Fig. S1) at the top, followed by a fractured layer

332 (R1) and fresh basement rock (R2). These results are coherent with those observed in other
333 basement contexts (Dewandel et al., 2006; Belle et al., 2019; Comte et al., 2012;). In the
334 sandstone and pelite domain, despite the complexity of the electrical pattern, the SR1
335 geological log distinguishes the alterite from the sandstone and pelite (C1, C2, Fig. S1) that
336 overlie a highly conductive C3 paragneiss derived from the sandstone and pelite, forming a
337 metamorphic halo (Egal et al., 1995) and being an aquifer formation. This suggests that the
338 high conductivity values observed are caused by strong and connected fracturing of the rock,
339 and thus that the paragneiss is the main aquifer formation of the catchment. The amphibolite
340 seems more impermeable than the sandstone and pelite, and the alterite forms a surface
341 aquifer (Figure 3). Further details are presented in the Supplementary material.

342 To complete the geological and geophysical investigations and improve preliminary
343 conclusions, chemical data were examined (Figure 4). The total mineralization of the water
344 can be assessed through its electrical conductivity (EC, $\mu\text{S}\cdot\text{cm}^{-1}$) and the Total Dissolved
345 Solids (TDS, $\text{mg}\cdot\text{L}^{-1}$), both parameters being correlated. The TDS values of the Spring
346 samples (natural outlet of the hydrogeological system) show small variations over the two
347 sampling periods (205-257 $\text{mg}\cdot\text{L}^{-1}$), as do most groundwater samples within the catchment
348 (182-235 $\text{mg}\cdot\text{L}^{-1}$, Pz1, Pz2, Pz3, Pz4, Pz7, Pz9 and Pz10). Nevertheless, four groundwater
349 samples (Pz6, Pz8, Pz12 and SR1) have clearly higher TDS values, ranging between 274
350 and 675 $\text{mg}\cdot\text{L}^{-1}$, varying both throughout the sampling periods and according to the sampling
351 depth (Figure 4B). Groundwater presents very heterogeneous chemical types at this small
352 catchment scale, from $\text{Ca}\cdot\text{HCO}_3$ to $\text{Ca}\cdot\text{SO}_4$. Cation proportions vary between a sodic end-
353 member (>50% Na+K, in Pz10) and a calcic end-member (up to 60% Ca), in Pz6 and Pz8
354 (Figure 4A).

355



356

357 *Figure 4 : (A) Piper diagram of the Plourhan groundwater samples. Both studied periods are shown, empty*
 358 *symbols are samples with data only available during the first period 2009-2011. (B) Total Dissolved Solids (TDS)*
 359 *versus Ca concentration.*

360 When comparing the revised structural scheme (Figure 3) and the geochemical data, it
 361 appears that samples with the higher calcium concentrations (i.e. Pz6, Pz8, Pz12 and SR1),
 362 which also have the highest TDS, reflect water-rock interactions with the weathered
 363 sandstone and pelite compartment. The other samples with low TDS values and lower
 364 calcium concentrations seems to results from water-rock interaction with the weathered
 365 amphibolite compartment. This chemical variability also reflects the complex geological
 366 settings of this small area (only 0.6 km²), in terms of both weathering profile and geological
 367 entities.

368 It is worth noting that water in the deep borehole SR1 has an increasing Ca load with depth,
 369 probably related to water rock-interaction with fractured sandstone. In addition, Pz4 and Pz7
 370 in the weathered sandstone and pelite compartment, present geochemical signatures very
 371 similar to that of groundwater in the amphibolite compartment. The geological and
 372 geophysical investigations allowed refining the contours of each geological formation on the
 373 1:50,000-scale geological map, these contours being coherent with the geochemical

374 groundwater signatures. However, interrogations remain for some piezometers that are too
375 far from the two geophysical profiles for making sufficiently reliable projections.

376

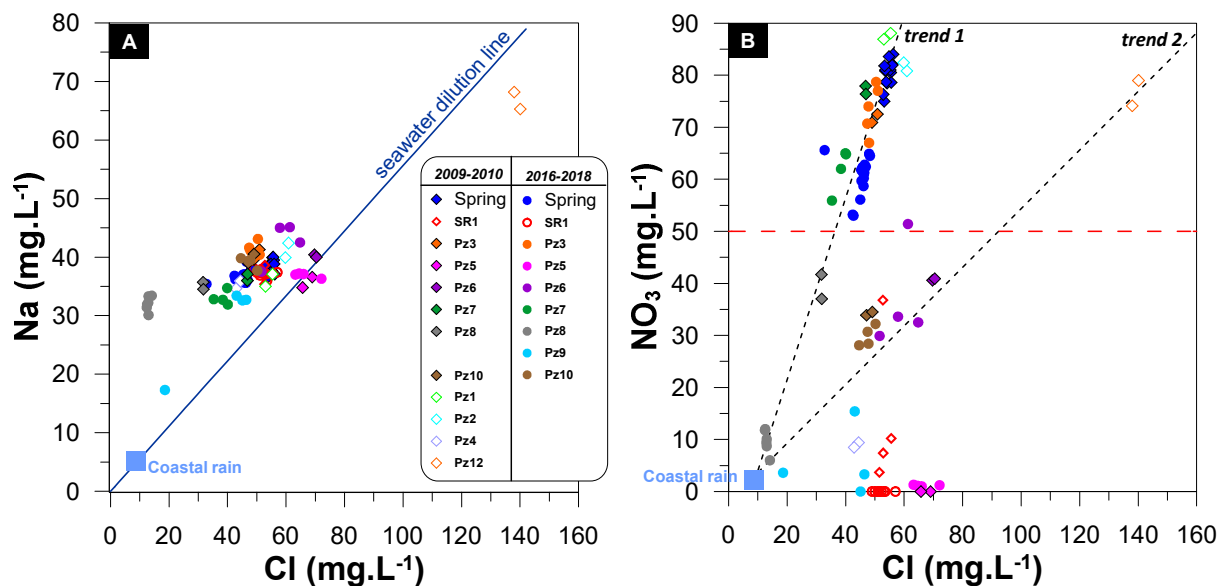
377 4.2 Temporal and spatial anthropogenic pressure: a focus on nitrate

378 As mentioned above, despite all efforts made to reduce nitrogen transfer to groundwater,
379 nitrate concentrations only decrease slowly at the Spring and remain above the 50 mg.L⁻¹
380 standard (Figure 2). The increase in NO₃ concentrations has stopped in the early 2000s,
381 reaching a plateau over five years before starting a decrease that continues today. This type
382 of evolution is typical of what is commonly observed in Europe since the implementation of
383 various programmes of measures for recovering a good chemical status of the groundwater
384 bodies, even though the response time can be quite different (European Commission, 2018).

385 This anthropogenic pressure is clearly shown in the small Plourhan catchment, where nitrate
386 concentrations in groundwater can be as high as 88 mg.L⁻¹ (Pz1, Oct-2009). However, the
387 nitrate concentrations in the piezometers sampled during the two sampling periods (2009-
388 2010 and 2016-2018), showed quite different values from one piezometer to the next. For
389 instance, the Spring and PZ7 showed a NO₃ decrease between the 1st and 2nd periods, from
390 80 to 61 mg.L⁻¹ (Spring) and from 77 to 62 mg.L⁻¹ (Pz7) (Figure 2; Figure 5B; table S1
391 Supplementary material). Pz8 has the largest decrease in nitrate concentrations, from 39 to
392 10 mg.L⁻¹ on average between the two sampling periods. The temporal variations between
393 1st and 2nd periods are less pronounced for Pz3 and Pz6. Most of the groundwater samples
394 from piezometers PZ5 and Pz9, and borehole SR1 now have nitrate concentrations below 10
395 mg.L⁻¹.

396 Figure 5A shows the sodium *versus* chloride concentrations in groundwater of the study
397 area. In the absence of evaporites in the local geological formations, chloride may derive
398 from atmospheric (Meybeck, 1983) or anthropogenic (agriculture, wastewater effluents, etc.)
399 inputs and thus may have the same origin as nitrate. Significant amounts of chloride (> 80
400 mg.L⁻¹) were measured in many groundwaters of the Brittany basement (> 80m deep) that
401 were shown to be related to the last marine Mio-Pliocene transgressions, while shallow and
402 modern groundwater has lower Cl concentrations (< 50 mg.L⁻¹) (Aquilina et al., 2015). In this
403 study, groundwater samples were taken at 10 to 22 m depth in the weathered layer and at
404 the top of the basement, except in the SR1 borehole that is 133 m deep. The Cl
405 concentrations are generally less than 60 mg.L⁻¹ including in borehole SR1. Only Pz12
406 presents a clearly higher Cl concentration of ~140 mg.L⁻¹ (two samples in 2009), as does Pz5
407 with Cl concentrations of 72 mg.L⁻¹ (the specific case of Pz5 is further discussed in section
408 4.3). For most groundwater samples, Figure 5A shows an excess of sodium relative to the

409 Na/Cl marine ratio, reflecting Na release in solution through interaction of groundwater with
 410 silicate minerals of the aquifer matrix. Only Pz12 has an excess of chloride compared to the
 411 Na/Cl ratio of the seawater dilution line, which may indicate anthropogenic input of Cl (both
 412 Na and Cl concentrations being higher than in the other samples). Manure and sewage
 413 effluents are known to be rich in Cl (Kloppmann, 2003), and KCl is used as fertilizers or road
 414 salts. The NO_3/Cl ratios of manure and sewage are generally lower than in mineral fertilizers,
 415 and can thus help distinguishing various anthropogenic sources (Lu et al., 2015). Moreover,
 416 the NO_3/Cl ratio may indicate a dilution effect *versus* nitrate removal through denitrification,
 417 as Cl behaves conservatively (Chen et al., 2009, Widory et al., 2013). Figure 5B shows NO_3
 418 vs. Cl concentrations, part of the groundwater samples with the highest nitrate
 419 concentrations (Spring, Pz7, Pz3, Pz2 and Pz1) defining a straight line correlating NO_3 and
 420 Cl concentrations (*trend 1*, Figure 5B), suggesting a common origin of both elements, i.e.
 421 that at least part of the Cl is anthropogenic. For the Spring and Pz7 that were sampled during
 422 both sampling campaigns, NO_3 and Cl concentrations decrease jointly between the two
 423 sampling periods, confirming this hypothesis. Pz12 with high NO_3 concentrations (75-80
 424 mg.L^{-1}) has a different NO_3/Cl ratio, with a higher Cl proportion. This may reflect a different
 425 anthropogenic source with a lower NO_3/Cl than previous samples, which could be related to
 426 manure input (*trend 2*, Figure 5B). Note that, as Pz12 is no longer accessible; it was not
 427 possible to resample this well during the 2nd period.



428
 429 *Figure 5 : (A) Sodium versus chloride concentrations in Plourhan groundwater samples. 'Coastal rain' refers to*
 430 *the mean rainfall at Brest (100 km to the west; Negrel et al., 2007). The Seawater dilution line (Bernier-Kay and*
 431 *Bernier, 1987) is presented as reference. Both studied periods are plotted, each sampling point has the same*
 432 *colour with different symbols: empty ones are samples from the first period (2009-2010). (B) Nitrate versus*
 433 *Chloride concentrations in Plourhan groundwater. The red dashed line represents the 50 mg.L⁻¹ nitrate standard.*
 434 *The black dashed lines represent the two main NO₃ vs. Cl trends.*

435 **Figure 5B** also shows groundwater samples that clearly differ from the previous group, with
436 similar Cl but lower NO₃ concentrations. Two sub-groups occur: a first one with NO₃ < 20
437 mg.L⁻¹ and Cl > 40 mg.L⁻¹ (Pz4, Pz5, Pz9, SR1), which may result from nitrate loss through
438 denitrification, or even no nitrate input. Note that Cl concentrations are in the range 40-75
439 mg.L⁻¹, except Pz9 sampled in April 2017 with 18 mg.L⁻¹. The second sub-group shows
440 intermediate nitrate concentrations (~30-50 mg.L⁻¹ in Pz6 and Pz10, and one SR1 sample at
441 60 m depth in November 2009), together with Cl ranging from 40 to 75 mg.L⁻¹. Such
442 characteristics may result from two distinct scenarios: (1) partial denitrification in water with
443 similar Cl and higher NO₃ concentrations (i.e. similar to the Spring signature); or (2) mixing
444 between a groundwater component with an anthropogenic impact similar to that described
445 for Pz12, and a groundwater component with low NO₃ and low Cl, typical of recharge by
446 rainwater. Finally, the Pz8 groundwater samples have NO₃ and Cl concentrations that
447 strongly decreased between the two sampling periods, in the same proportions that were
448 observed for the Spring group. Moreover, the 2018 samples, which have similar Cl
449 concentrations, have NO₃ concentrations that decrease with the sampling depth, reflecting a
450 stratification of the local groundwater.

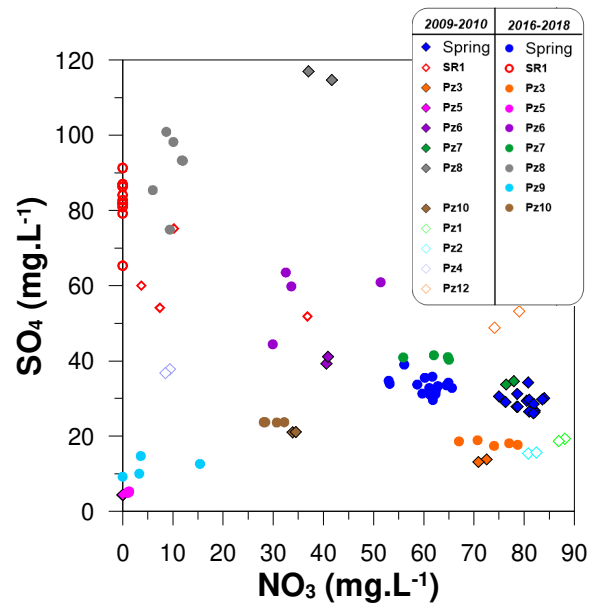
451 To summarize, the joint decrease of NO₃ and Cl concentrations between the two sampling
452 periods could result from a positive impact of the mitigation measures applied on the
453 catchment since about two decades, while a net decrease of nitrate alone should result from
454 a chemical/microbiological process like denitrification.

455

456 4.3 Evidence of denitrification processes based on N and O isotopes of 457 NO₃, microbiology and molecular approaches

458 Denitrification leads to a decrease, or even 'disappearance' (below the analytical
459 quantification limit), of nitrate concentrations in groundwater. Denitrification by oxidation of
460 pyrite has been identified as an important denitrification pathway in the Brittany basement-
461 rock aquifers, where pyrite is abundant (Pauwels et al., 2000). Pyrite oxidation releases
462 dissolved sulphate in solution. Except for samples Pz5 and Pz9, discussed later, **Figure 6**
463 shows an overall trend of increasing sulphate content when the nitrate content decreases,
464 which is typically the case for SR1 samples; Pz8 samples were sampled at various depths
465 and, over both sampling periods, they had the highest SO₄ concentrations and medium to
466 low NO₃ concentrations. Even Pz6 samples could indicate denitrification through pyrite
467 oxidation. Note that pyrite was observed in fractures of the sandstone and pelite during
468 drilling of Pz6 and Pz8. In the Spring and Pz7 samples, while nitrate concentrations dropped

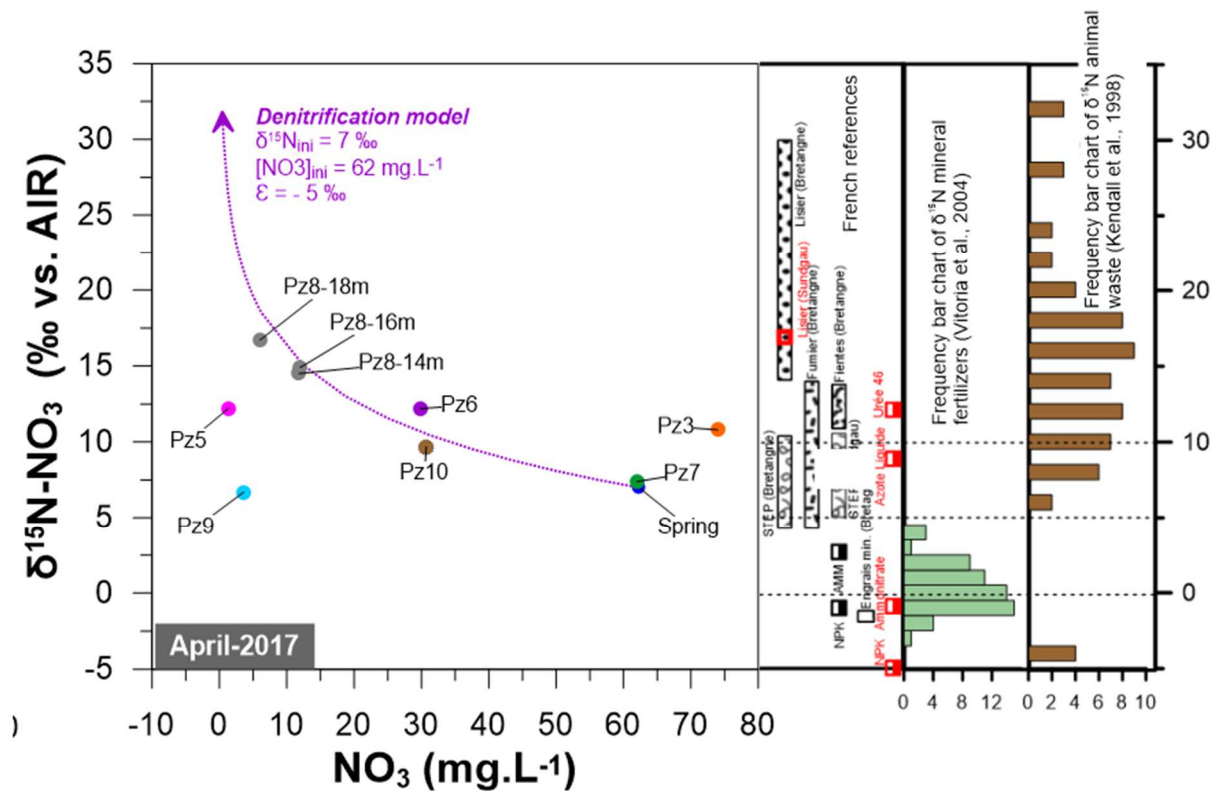
469 significantly between the two sampling periods, there was no significant increase in sulphate
470 contents.



471

472 *Figure 6 : Sulphate versus nitrate concentrations in mg.L⁻¹ in Plourhan groundwater samples for both sampling*
473 *periods.*

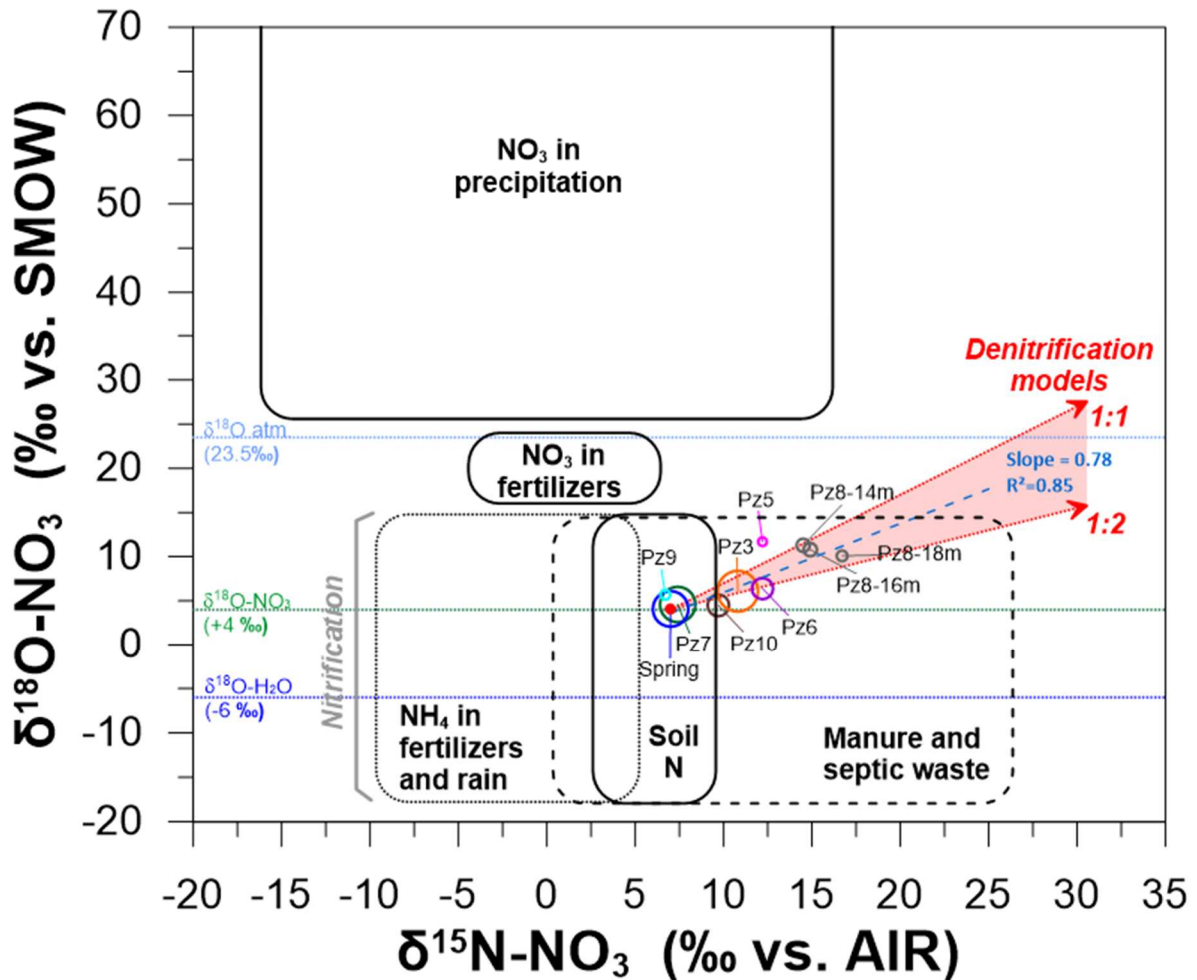
474 The dual approach of N and O isotopes of nitrate was applied to a selection of samples from
475 each major group defined above (Figure 5B). Among these samples from April 2017, ten
476 were analysed to determine their NO₃ isotopic compositions (Table S2, Supplementary
477 material). Note that the three samples from borehole SR1 could not be isotopically analysed
478 because nitrate concentrations were below the quantification limit. The δ¹⁵N-NO₃ isotopic
479 ratios of all samples ranged from +7 to +17‰, with nitrate concentrations ranging from 1 to
480 74 mg.L⁻¹ in Pz5 and Pz3, respectively. For the samples with the highest nitrate
481 concentrations, the Spring and Pz7 had a δ¹⁵N-NO₃ of ~7-7.5 ‰ that could reflect a mixed
482 influence of mineral and organic fertilizers, while Pz3 with a δ¹⁵N-NO₃ of ~10.8 ‰ appears to
483 be more influenced by organic fertilizer nitrate input (Figure 7), both types of fertilizer being
484 used in the catchment.
485



486

487 *Figure 7 : $\delta^{15}\text{N-NO}_3$ vs. NO_3 concentrations for the April 2017 samples. The purple dashed line represents the*
 488 *denitrification model according to the Raleigh equation, starting from the Spring sample characteristics. Bar charts*
 489 *on the right side represent the $\delta^{15}\text{N}$ frequencies of mineral fertilizers and animal waste. The main nitrate sources*
 490 *measured in France are also shown (Kloppmann et al., 2003; Widory et al., 2005).*

491 **Figure 7** also shows that, with the exception of Pz5 and Pz9, the samples present an
 492 increasing $\delta^{15}\text{N-NO}_3$ when NO_3 concentrations decrease, which is a typical trend of
 493 denitrification according to the Rayleigh model. Here, we chose to apply a denitrification
 494 model starting from the sample signatures with the highest nitrate concentration, and
 495 assumed not to be impacted by denitrification as shown by their chemical composition. The
 496 denitrification model is thus based on the signature of Spring and Pz7 samples ($\delta^{15}\text{N-NO}_3 = -$
 497 7‰ , $[\text{NO}_3] = 62 \text{ mg.L}^{-1}$) with an enrichment factor $\epsilon = -5\text{‰}$. It explains a partial denitrification
 498 for samples Pz6, Pz10 and Pz8 at different depths as these samples plot along the
 499 calculated denitrification curve (Figure 7). The relatively low ϵ value (-5‰) is similar to the
 500 one found by Pauwels et al. (2000) in the deep aquifer of the pyrite-bearing Brioverian schist
 501 at the Naizin site (SE Brittany), and suggests a relatively rapid denitrification process
 502 (Mariotti et al., 1988) in full agreement with the results of Pauwels et al (1998) in the same
 503 area. Note that a low ϵ value (-4‰) was further confirmed in the same site during an artificial
 504 tracer test along a fracture in the pyrite-bearing Brioverian schist basement (Petelet-Giraud
 505 et al., 2017).



506

507 *Figure 8: $\delta^{18}\text{O-NO}_3$ versus $\delta^{15}\text{N-NO}_3$ with typical $\delta^{15}\text{N}$ and $\delta^{18}\text{O}$ values of nitrates derived or nitrified from various*
 508 *nitrogen sources (after Kendall, 1998). The two red "denitrification" arrows indicate typical slopes expected for*
 509 *data resulting from nitrate denitrification with initial $\delta^{15}\text{N} = +7\text{‰}$ and $\delta^{18}\text{O} = +4\text{‰}$. Typical ranges of values for*
 510 *$\delta^{18}\text{O-NO}_3$ produced by nitrification of ammonium and organic matter are referred to as "Nitrification". The size of*
 511 *the circles for each sample of April 2017 is proportional to the measured nitrate concentration.*

512 All isotopic data of the April 2017 sampling campaign are shown in Figure 8, coupling $\delta^{15}\text{N-NO}_3$
 513 NO_3 and $\delta^{18}\text{O-NO}_3$. Firstly, it should be noted that the original $\delta^{18}\text{O-NO}_3$ isotope signature of
 514 synthetic mineral fertilizers ($\delta^{18}\text{O-NO}_3 \sim +20\text{‰}$) is not found in groundwater samples, even
 515 before any denitrification (Spring and Pz7 samples). The initial ^{18}O isotopic signature seems
 516 to have been lost due to immobilization and mineralization processes of nitrogen in the soil
 517 (Mengis et al., 2001, Sebilo et al., 2013). Among the three oxygen atoms, associated with
 518 nitrogen of the newly formed NO_3 , two come from local water and one from atmospheric O_2
 519 ($\delta^{18}\text{O-O}_2 \sim +23\text{‰}$; Equation 1). The isotopic signature of the local groundwater can be
 520 approximated using the mean isotopic signature of the local rainfall (weighted average of the
 521 precipitation amounts). According to the French national map by Millot et al. (2010), the local
 522 groundwater signature is estimated to be $\delta^{18}\text{O-H}_2\text{O} = -6\text{‰}$. According to Equation 1, the
 523 theoretical $\delta^{18}\text{O}$ value of the newly formed nitrate in the soil is thus $\delta^{18}\text{O-NO}_3 \sim +4\text{‰}$, which

524 agrees with the measured values of the Spring and Pz7 samples (+4.1 and +4.6 ‰
525 respectively, and considering an analytical accuracy of ± 0.5 ‰).

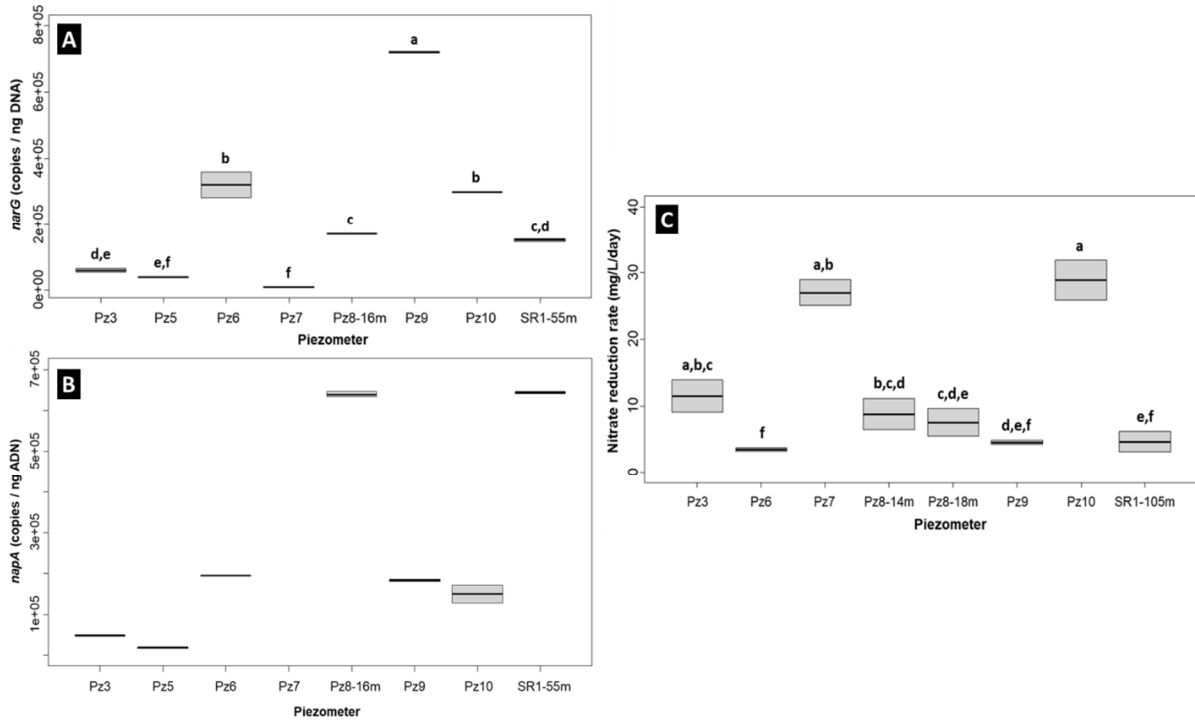
526 The dual $\delta^{15}\text{N-NO}_3$ and $\delta^{18}\text{O-NO}_3$ isotopic approach confirms a denitrification process for
527 several samples, with a joint increase in the $\delta^{15}\text{N}$ and $\delta^{18}\text{O}$ isotopic ratios of the residual
528 nitrate (Figure 8). The slope of the relation between $\delta^{18}\text{O-NO}_3$ and $\delta^{15}\text{N-NO}_3$ for the samples
529 undergoing denitrification is 0.78 ($R^2=0.85$), indicative of denitrification activity (Aravena and
530 Robertson, 1998). In more details, samples can plot along a 0.5 slope (e.g., Pz6) and along a
531 1 slope (e.g., Pz8-14). Compilation of a large number of environmental studies showed that
532 denitrification increases $\delta^{18}\text{O}$ and $\delta^{15}\text{N}$ in a ratio of about 0.5; which corresponds to a $^{15}\text{N}\epsilon$
533 about twice as large as $^{18}\text{O}\epsilon$ (Mengis et al., 1999). A lower slope between $\delta^{18}\text{O-NO}_3$ and $\delta^{15}\text{N-NO}_3$
534 of nearly 0.5 was also reported from field studies, and another of nearly 1.0 from
535 laboratory studies (Otero et al., 2009; Critchley et al., 2014; Margalef-Marti et al., 2019;
536 Ceballos et al., 2020). The main hypothesis now recognized for this flatter slope in field-data
537 sets compared to that in laboratory data, is the oxidation of the intermediates products (NO_2
538 and/or NH_4) to NO_3 (Granger and Wankel, 2016; Margalef-Marti et al., 2019), while a direct
539 NO_2 reduction to gaseous N products might lead to values close to the theoretical 1:1 line.
540 Thus, it is generally accepted that when isotopic nitrate data plot along a slope from 0.5 to 1,
541 the trend can be interpreted as a denitrification process.

542 Samples Pz6, Pz10 and the three Pz8 samples (14, 16 and 18m deep) are clearly marked by
543 a denitrification, with an alignment describing a slope between 0.5 and 1 (Figure 8). In
544 addition, Pz6 and Pz10 samples show dissolved N_2O (intermediate denitrification product),
545 depending on the sampling campaigns, at concentrations that can reach $200 \mu\text{g.L}^{-1}$. It should
546 be noted that Pz3, although in the same alignment, is not affected by denitrification, but
547 rather by a different source of nitrate (more “organic” in this case, Figure 7 and Figure 8) with
548 the highest NO_3 concentration measured during the study. Pz5 and Pz9, with low nitrate
549 contents (and low sulphate contents) and have $\delta^{15}\text{N-NO}_3$ isotope signatures that do not
550 indicate any denitrification.

551 Isotopic analyses thus suggested a microbial nitrate reduction activity in some piezometers,
552 i.e. Pz6, Pz8 and Pz10, but not all the piezometers in the catchment. In water samples of
553 2017, the relative abundance of copies of *narG* and *napA* genes, encoding nitrate reductases
554 involved in nitrate reduction by denitrifying bacteria, was higher in Pz6, Pz8 and Pz10 than in
555 other piezometers, such as Pz3, Pz5 and Pz7, for which isotopic investigation suggested no
556 nitrate reduction (Figure 9 A and B). Results obtained from isotopic investigation and
557 molecular analyses agree, showing higher nitrate-reductase gene abundances in
558 piezometers where nitrate reduction was highlighted by isotopic signatures. Abundance of

559 gene copies was also assessed over a period of two years (through the four sampling
560 campaigns including April 2017), revealing the presence of both *narG* and *napA* genes in all
561 piezometers of the catchment, thus showing a potential for microbial nitrate reduction in all
562 studied piezometers of the Plourhan site (data not shown). Spatio-temporal variations in
563 gene abundance were highlighted according to the sampling time and location. This agrees
564 with earlier results that showed a temporal variation in the abundance of denitrification genes
565 in groundwater, and a wide-ranging spatial distribution of these genes in a single catchment
566 ([Hernandez-del Amo et al., 2018](#)).

567 To complete the isotopic and molecular approaches for nitrate reduction characterization in
568 samples collected in April 2017, the potential nitrate reduction activity of microbial
569 communities sampled from several piezometers was measured in batch experiments, run
570 under standardized conditions for denitrification. Nitrate reduction activity was found in all
571 the tested wells (Figure 9C). This agrees with the quantification of functional genes that were
572 also detected, albeit at different relative abundances, in all piezometers (Figure 9 A and B),
573 showing a spatially widespread potential of the groundwater microbial communities for
574 reducing nitrate. These results agree with earlier work, showing the wide distribution of
575 microbial communities involved in denitrification in groundwater ([Hernandez-del Amo et al.,
576 2018](#)). Batch experiments showed a high heterogeneity among the piezometers, as nitrate
577 reduction rates can be high (Pz7 and Pz10) or low (Pz6 and Pz9). They also showed that the
578 complete denitrification pathway occurred, as nitrite (NO_2^-) and N_2O were detected as well
579 (data not shown). This strongly suggested that a complete denitrification can occur in the
580 different groundwater of the catchment provided that *in situ* conditions are favourable.



581
 582 *Figure 9 : Microbiological and biomolecular analyses of water collected in piezometers at the Plourhan site in April*
 583 *2017. A and B: Relative abundance of narG gene copies (A) and napA gene copies (B) involved in bacterial*
 584 *nitrate reduction, determined with quantitative PCR. (C) Potential nitrate reduction activity measured in batch*
 585 *experiments under standardized conditions for denitrification (for all piezometers, no nitrate reduction was*
 586 *observed in abiotic flasks). Significant differences between conditions were searched for by means of the Kruskal-*
 587 *Wallis non-parametric test ($p>0.05$) and are mentioned as letters (a, b... f).*

588 Intermediate and high potential nitrate reduction rates found in batch experiments for Pz8 (at
 589 two depths) and Pz10, respectively, agree with the denitrification activity showed by isotopic
 590 data and the *narG* and *napA* genes abundance results found for these piezometers. The low
 591 potential nitrate reductase activities found for Pz9 and SR1 also agree with the isotopic
 592 results, suggesting no denitrification for Pz9. The results were inconclusive for SR1 as the
 593 nitrate concentration was too low, and because of the low abundance of genes encoding the
 594 nitrate reductase enzymes. The results from the batch experiments for these four waters thus
 595 agreed with the isotopic and molecular approaches.

596 However, no relation was found for the other piezometers. For Pz3 and Pz7, intermediate to
 597 high activity was measured in batch experiments, but isotopic and molecular approaches
 598 both suggested a low nitrate reduction potential. For Pz6, whereas the isotopic and
 599 molecular approaches suggested nitrate reduction activity and a potential for such activity,
 600 respectively, batch experiments only showed a weak nitrate reduction potential. These
 601 differences could be due to unsuitable *in situ* conditions for denitrification compared to batch
 602 experiments, in the case where batch experiments showed a high potential, but no nitrate
 603 reduction was detected using isotopes. Another explanation might be that the *in situ* nitrate

604 reduction, shown by isotopic work is actually ancient, explaining why only a weak activity was
605 detected in batch experiments.

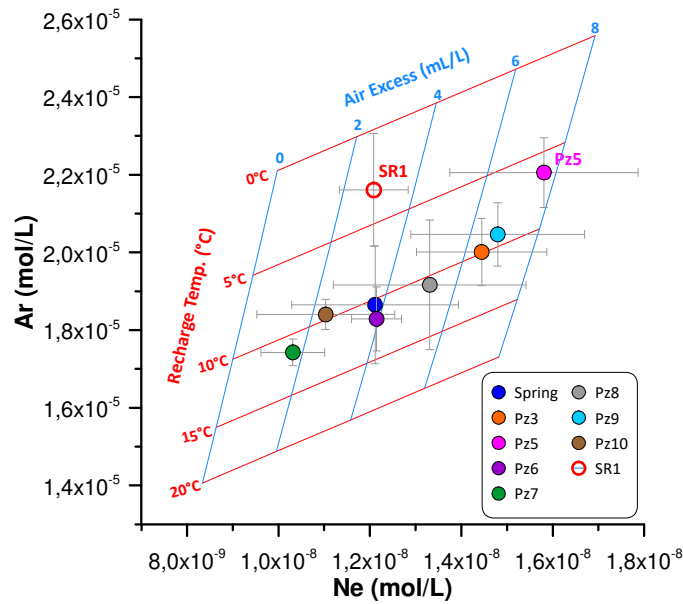
606 Therefore, according to geochemistry, isotopic tools and/or biomolecular characterization,
607 the Pz5, PZ9 and SR1 wells highlight different water origins and/or mechanisms than in other
608 piezometers. This is further investigated in the following section.

609

610 4.4 Groundwater residence time assessment: CFCs, SF₆ and noble 611 dissolved gases

612 Dissolved gases (CFCs and SF₆) were analysed during the two sampling periods (2009-2011
613 and 2016-2018), to assess the groundwater residence time and the local aquifer dynamics
614 (Table S3, Supplementary material). In this type of aquifer, the exponential model—reflecting
615 a diffuse recharge at the catchment scale resulting in a mixture of old and young
616 groundwater components at the outlet—generally is the most appropriate one. Here,
617 however, the binary mixing model, that mixes different reservoirs, could be also suitable for
618 reflecting the structure of the basement aquifer, consisting of fractured bedrock overlain by a
619 weathered layer.

620 The Spring samples appear to be systematically contaminated with CFC-11 and CFC-12.
621 However, as CFCs have been used worldwide in a wide range of activities, this type of
622 contamination is common near urban areas or landfills (Höhener et al., 2003) and both
623 tracers are unsuitable for dating purposes. Only CFC-113 and SF₆ can be used, thus
624 increasing the uncertainty associated with estimating a mean residence time. According to
625 the chemical characteristics of the Spring water and its relative stability during the year,
626 groundwater dating tracers are interpreted with exponential model. Based on these two
627 tracers, the mean τ value (characteristic time needed for the two-thirds renewal of the aquifer
628 water according to the exponential model) of the Spring waters is about 20 ± 5 years. For the
629 other piezometers (Pz) and borehole (SR1) in the catchment area, different groups can be
630 used.



631
 632 *Figure 10: Mean Neon versus Argon concentrations (mol/L) for each groundwater sample monitored during the*
 633 *period 2016-2018 (n=2 for Pz10 to n=16 for the Spring). Blue and red lines represent the theoretical values from*
 634 *solubility equations for an altitude of 100 m, a temperature between 0 to 20 °C and an excess air between 0 and 8*
 635 *mL/L. Samples plotting in around the same red line have the same recharge temperature, but different excess air.*

636 A first group of samples is marked by lower recharge temperatures than the present-day one
 637 of 11-12°C in Brittany (Ayraud et al., 2008). The noble-gas concentrations (Ar and Ne, Figure
 638 10) measured in the SR1 borehole showed a recharge temperature of around 3-4 °C, i.e. 7 to
 639 9 °C lower than the present-day one, indicating the presence of old groundwater in the
 640 system that could be related to the last glaciation period of -19 to -17 ky (Aquilina et al.,
 641 2015). In addition, the CFC and SF₆ measurements also indicate a long residence time
 642 (mean 230±40 yr using an exponential model), with a possible contribution of a recent
 643 component not exceeding 10 % that may reflect water inflow resulting from the pumping for
 644 groundwater sampling (Figure 11).

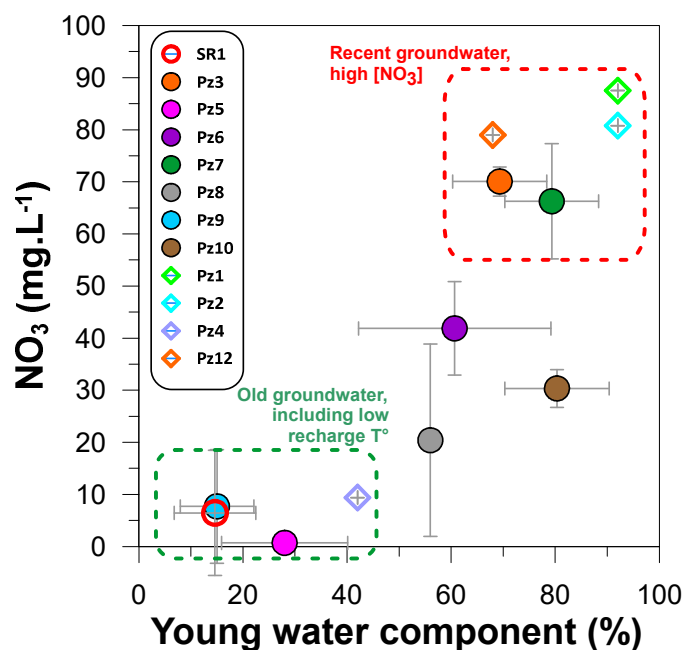
645 This recent component contained nitrate in some SR1 samples from the first sampling period
 646 (2009), with NO₃ reaching 38 mg.L⁻¹ at 60 m depth versus 3.7 mg.L⁻¹ at 110 m depth. In
 647 addition, during the second sampling period (2017-2018), an excess of 50 to 100% dissolved
 648 N₂ was determined compared to Ne and Ar concentrations, with concentrations reaching
 649 1.2.10⁻³ to 1.4.10⁻³ mol.L⁻¹ compared to 7.10⁻⁴ to 8.10⁻⁴ mol.L⁻¹ in the Spring samples This
 650 may be a result from the denitrification of the incoming nitrate, no NO₃ being detected (<LQ)
 651 during the second sampling period. In a second location, Pz5 tapping the saprolite and the
 652 upper part of the fractured amphibolite, noble-gas analyses (n=2, Figure 10) also showed a
 653 low recharge temperature of around 6 °C. While no nitrate was detected during the first
 654 sampling period, about 1 mg.L⁻¹ was measured during the second sampling period, which
 655 could corroborate the CFC-SF₆ analyses indicating between 70 and 85 % of old water (i.e.

656 CFC-SF₆ free), and thus a small contribution of recent water possibly enriched in nitrate
657 (Figure 11).

658 Above, we also pointed out a relatively high Cl concentration of up to 72 mg.L⁻¹, i.e. the
659 highest concentration in the studied area (except Pz12 with both high Cl and NO₃
660 concentrations). An anthropogenic Cl contribution is unlikely as there is no nitrate and no
661 denitrification process was detected. Pz5 samples are the only ones with a Na/Cl ratio close
662 to that of seawater (Figure 5A). Combining the low recharge temperature, the absence of
663 nitrate and the relatively high Cl concentration considered to be natural, this sampling point
664 could reflect the legacy of marine transgressions over the past 5 millions years. Armandine
665 Les Landes et al. (2015) considered that Cl of marine origin, not supplied by precipitation (>
666 40 mg L⁻¹) is likely related to marine transgressions that flooded the Armorican Massif since
667 the Messinian. The large quantities of Cl injected during each transgression were then
668 diluted by meteoric recharge after the end of the transgression period. This includes
669 recharge during the last glaciation, shown by a low recharge temperature indicated by noble
670 gases. In Pz5 at 18 m depth, the water comes from the upper part of the amphibolite
671 basement and from the lower part of the overlying weathered material. This old water,
672 preserved from anthropogenic impact, is disconnected from the groundwater feeding the
673 Spring about 500 m distant, relatively disconnected from present day recharge, and only
674 remobilized by pumping for sampling.

675 A second group, represented by Pz9, has a long mean residence time of over 60 years
676 (Figure 11) , with a large proportion (80-90%) of old water and no CFC-SF₆, but no evidence
677 of low recharge temperatures. These waters thus appear to be younger than the ones of Pz5
678 and SR1, but old enough not to have received anthropogenic nitrate.

679 The other groundwater samples have apparent ages of about 30 years, such as Pz10 and
680 Pz3. Pz6 shows a variability between the two samples analysed for dating purposes. This
681 can reflect mixing in various proportions between old and present-day waters (Figure 11)
682 according to the period of the hydrological cycle, the apparent age being slightly older in May
683 2018 than in October 2017. This variability was also observed for the major elements, which
684 is typical of the sandstone and pelite aquifer that has different aquifer dynamics than the
685 amphibolite aquifer. Pz7 has a high SF₆ contamination; based on the other tracers, the
686 waters could correspond to a mixing between a large proportion of recent water (70-80%)
687 and a 30-40-years old water.



688

689 *Figure 11: Mean NO₃ concentrations versus the mean percentage of the young-water component in each*
 690 *groundwater sample of the study area.*

691 Such different groundwater residence times are known from other parts of Brittany.
 692 Evidence of past climatic and geological event was found in groundwater from Lanleff
 693 monitoring station < 10km west of Plourhan (Armandine Les Landes et al., 2015) and in
 694 various other parts of Brittany by Aquilina et al. (2015). Most shallow groundwaters at a
 695 regional scale, however, is strongly impacted by anthropogenic activity and has mean
 696 apparent residence times of 10 to 30 years (Ayraud et al., 2008). All earlier studies studied
 697 different regions, which raises questions about the functioning and connectivity of the aquifer
 698 systems studied, questions that might be answered by detailed geological studies of the
 699 sites.

700

701 5 Conclusions and perspectives

702

703 The Plourhan catchment (~100 ha) contains a main spring that still exceeds the limit of 50
 704 mg/L NO₃ despite the numerous remedial measures implemented since the 1990s. We
 705 investigated the catchment with complementary and robust geological and hydrogeological
 706 approaches, better to constrain the driving parameters of nitrate transfer and distribution in
 707 groundwater, and thus to improve our understanding of how the hydrosystem functions.

708 First, geochemical water sampling over about 10 years showed a quite high variability in
 709 groundwater quality: (1) a temporal variability especially regarding nitrate and (2) a spatial
 710 variability despite the small size of the catchment. Though a link with agricultural practices

711 and land use is one of the key factors controlling nitrate fate in groundwater, other factors
712 must be considered as well, such as the hydro-biochemical functioning of the aquifer system,
713 which can also largely control water quality.

714 Secondly, the study of nitrate isotopes (N and O) clearly demonstrated denitrification in
715 some piezometers of the catchment area. Such processes were generally demonstrated by
716 microbiology and molecular biology studies, highlighting the presence of functional genes
717 involved in denitrification as well as the capacity of the groundwater microbial community for
718 denitrifying when *in situ* anoxic and/or nutritive conditions are favourable. Other groundwater
719 samples had low nitrate contents, even below the quantification limit, without any
720 denitrification process occurring. Study of the average groundwater residence time, based on
721 dissolved CFCs and SF₆ and on rare gases; locally discovered old waters, particularly in
722 nitrate-free piezometers, demonstrating that such waters never received any anthropogenic
723 nitrate. One piezometer contained old water, with a recharge temperature well below the
724 current average temperature in the area, reflecting water from the last glaciation period (-19
725 to -17 ky).

726 Third, the presence of such very old groundwater in the catchment area, raises questions
727 about the structure of the aquifer system at the scale of the recharge area. The detailed
728 geological and geophysical work, especially with electric tomography profiles, led to
729 improvement of the local structural model. A large number of compartments delimited by
730 faults was shown to exist, as was a highly variable thickness of soil and weathered rock in
731 the study area. This better understanding of the geological structure has shown the
732 heterogeneity of the hydrosystem, where some compartments appear to be disconnected
733 from the general groundwater flow. This may explain the presence of old residual
734 groundwater not affected by the agricultural activity in the catchment.

735 The Plourhan site has shown the relevance and necessity of using an integrated approach
736 for significantly improving our knowledge in terms of the functioning of complex
737 hydrosystems—such as basement ones— and its impact on groundwater quality. This
738 approach can be advantageously applied to other sites, especially in basement domains
739 where denitrification processes can deeply modify the local water quality, and where the
740 geological structure may be quite complex. The various methods of such an approach can
741 be implemented independently, and successively, steered by gradually improving
742 knowledge, or in a coupled manner as done here. The understanding of hydrodynamic
743 functioning *sensu lato* is as important as that of nitrate reduction. Both are paramount for
744 implementing the most relevant programmes of measures (as required by the European
745 Water Framework Directive; Fifth WFD and FD Implementation Reports, 2019), for

746 recovering a good water quality and assessing the effectiveness of such measures and
747 methods, notably by considering the timeframe between implementation of the measures
748 and their impact on groundwater quality.

749

750 **6 Acknowledgments**

751 The French Loire-Bretagne Water Agency (AELB) together with the BRGM financially
752 supported this study through the POLDIF Research project led by N. Baran. The authors
753 thanks Florian Koch, Benjamin Maurice and Mickael Charron for their precious assistance in
754 the field, Marine Gremont for her synthesis of the measures for recovering water quality. All
755 colleagues from the BRGM Labs involved in the data production are thanked for their
756 valuable work. Jean-Michel Schroetter is warmly thanked for the many fruitful discussions on
757 the local geological settings and how they can govern water circulations and water quality. E.
758 Petelet-Giraud addresses special thanks to Gilles L. and Christophe G. for their fruitful
759 comments on figure 8. We are grateful to Dr. H.M. Kluijver for proofreading and editing the
760 English text.

761

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