# Do macroinvertebrate abundance and community structure depend on the quality of ponds located in peri-urban areas?

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## Abstract

Contamination is one of the major threats to freshwater biodiversity. Compared to other aquatic ecosystems, peri-urban ponds are unique because they are embedded in human-dominated areas. However, it is poorly understood how different land uses such as urban or agricultural contribute multiple pollutants to ponds and thus affect pond biodiversity. In this work, 12 ponds located in a peri-urban area (Ile-de-France region, France) were monitored for 2 consecutive years in spring and fall. We surveyed macroinvertebrates and measured the physicochemical parameters and contaminants of different classes (trace elements, pharmaceuticals, pesticides and polycyclic aromatic hydrocarbons) in both water and sediment. The objective was twofold: (1) to explore local and regional macroinvertebrate spatiotemporal diversity and (2) to understand the effects of contaminants on community structure. We observed 236 macroinvertebrate morphotaxa, none of which were rare or sensitive to pollutants. Morphotaxa richness showed small differences between ponds but no difference between there was no effect of field campaign. There was no effect of ponds and field campaign on morphotaxa diversity and equitability. We did not observe a relationship between land use around the pond (agricultural, urban, or semi-natural) and diversity indices with the exception of the proportion of agricultural land in the vicinity of the pond on equitability. Regional beta diversity (between ponds) showed that differences in morphotaxa composition reflected species replacement more than differences in species richness; these were primarily due to the high abundance of pollutant-tolerant species in some of the ponds. The effects of environmental parameters on community structure were studied using partial redundancy analysis based on the presence-absence of morphotaxa, showing that community assemblages are shaped by sediment levels of pharmaceuticals, water conductivity and ammonium concentration. In conclusion, ponds in peri-urban areas are exposed to various human activities, with our results suggesting that this exposure leads to chronic and diverse contaminations that affect morphotaxa communities.

**Keywords**: beta diversity, land use, pollutant-tolerant morphotaxa, trace elements, PAH, pharmaceuticals, herbicides, insecticides, fungicides

## 1. Introduction

Freshwater ecosystems, including lakes, reservoirs and rivers, host around 9.5% of all described species despite representing approximately 2.3% of the world's surface area and only 0.01% of its water (Reid et al. 2019). They therefore deserve special attention in terms of their protection and conservation. In a recent review, Reid et al. (2019) listed 12 emerging or intensifying threats to freshwater biodiversity. Among the different freshwater ecosystems, ponds are particularly vulnerable because of their small size (Biggs et al. 2017), although they are understudied probably on account of their presumed insignificance (Biggs et al. 2017; Cereghino et al. 2008). Several studies showed that small water bodies such as ponds, ditches and streams host a higher biodiversity than large water bodies (Biggs et al. 2017). In particular, because of their isolation or small size, they may have low local alpha diversity. However, a high dissimilarity between these aquatic systems may lead to high regional beta diversity (Clarke et al. 2008; Davies et al. 2008; Williams et al. 2004). The small habitat size of ponds may also support rare and scarce species. Scheffer et al. (2006) showed that shallow lakes and ponds support a higher richness of aquatic birds, plants, amphibians and invertebrates than large water bodies. This higher richness is accompanied by a low diversity of fish, if not their complete absence (Scheffer et al. 2006).

Peri-urban areas are characterised by complex landscapes with both agricultural and urban covers and a mixture of different uses and users (Poggi et al. 2021; Zoomers et al. 2017). They are typical zones of continuous transformation (Zoomers et al. 2017) or "restless landscape" (Friedmann 2016). However, ponds and wetlands in peri-urban environments are understudied (Wanek et al. 2021). Although urban ponds are less diverse than rural ponds, they may host threatened species, thus advocating for good management practices (Oertli

and Parris 2019). Likewise, motorway stormwater retention ponds can play a significant role in macroinvertebrate diversity at the regional level (Le Viol et al. 2009; Meland et al. 2020).

Contamination by pollutants is one of the 12 threats to freshwater biodiversity identified by Reid et al. (2019). Ponds are subjected to multiple contaminations depending on their environment, whether in an agricultural, urban or highway setting. These stressors have been studied both independently and in combination and include, among others, pesticides (Trigal et al. 2007), major ions, nutrients and wastewater-associated micropollutants (Berger et al. 2018), polycyclic aromatic hydrocarbons (PAH) (Uher et al. 2016), pharmaceuticals and heavy metals (Andreu et al. 2016), metals, ions and PAH in combination (Sun et al. 2018). The distinctive feature of peri-urban ponds is that they are embedded in a human-dominated matrix with different activities, which can be the source of multiple stressors. In peri-urban areas, although ponds are not located far from each other, they might be exposed to different dominant stressors or combine different categories of pollutants because of their small catchment area (Biggs et al. 2017; Cereghino et al. 2008). In addition, contaminants in ponds may change with time if the surrounding landscape is submitted to changes such as urbanisation or temporary construction sites, or if it is exposed to new materials such as nanomaterials or personal care product additives. Ponds may accumulate these contaminants, particularly in their sediments and, consequently, freshwater life is affected by changing contaminant cocktails.

Aquatic macroinvertebrates encompass a rich and diversified set of taxa that are universally found in freshwater ecosystems. They exhibit a wide range of sensitivity to environmental stressors and, as a consequence, their local diversity and abundance are commonly used as

indicators of perturbations (Sumudumali and Jayawardana 2021; Tachet et al. 2010). Aquatic macroinvertebrates are mostly sedentary, at least during their larvae stage. They inhabit different habitats and their life cycle, for the majority of macroinvertebrates, is annual (Tachet et al. 2010). For these reasons, these organisms are good indicators of pollution, as the recolonisation of perturbed areas takes time. Macroinvertebrate indices of water quality are based on the presence-absence or abundances of macroinvertebrates. However, in addition to the local diversity known as alpha diversity, spatial and temporal diversity give information respectively about the spatial structuration and temporal changes of communities (Legendre and Condit 2019). Thus, their spatial and temporal beta diversity might prove useful to assess the impacts of stressors in changing peri-urban areas.

Here we report the results of a study in which we monitored 12 ponds in a peri-urban area located in the Ile-de-France region (France). The chosen ponds were characterised by different proportions of agricultural, urbanised, grassland and forest surfaces in a 100-m radius buffer. We therefore aimed to link land use, contaminant concentrations in water and sediment as well as macroinvertebrate distribution in ponds. Our objective was to understand whether water and sediment pollutants and land-use characteristics are constraints to macroinvertebrate distribution in ponds. To do so, we sampled macroinvertebrates and measured pond quality parameters, including various urban and agricultural contaminants, to quantify the main pesticides, pharmaceuticals, PAH and trace elements (TE) in water and sediments. Depending on their chemical properties, contaminants may accumulate either in water or in sediment. We therefore monitored both compartments. Nélieu et al. (2020) showed that the water contamination profiles of these ponds differed depending on their location, and that the agricultural landscape explained these differences more than urban land

uses. In some of the ponds, the environmental risk exceeded the thresholds of risk quotient mainly due to pesticides (Nélieu et al. 2020). Here, we analysed the macroinvertebrate communities and hypothesised that (1) local macroinvertebrate diversity is higher in ponds located in environments dominated by grasslands and forests than in those dominated by agriculture and urban areas; (2) ponds hosting rare and pollutant-sensitive macroinvertebrate morphotaxa highly contribute to regional diversity; and (3) water and sediment contaminants influence morphotaxa distribution in ponds.

## 2. Material and methods

### 2.1. Study area

The selected study area is the Saclay Plateau (N: 48° 43' 59.99" E: 2° 10' 0.01), located in the junction zone between the Parisian agglomeration and its large surrounding plains. Until recently, the territory had a mainly agricultural vocation, although the ongoing development of a scientific and technological pole in the area has considerably increased the urban hold on the territory. At the same time, the Natural, Agricultural and Forest Protection Zone of the Saclay Plateau extending over more than 4,000 ha was created in 2010, thus perpetuating the agricultural use of land on the plateau. The Saclay Plateau thus presents major challenges in terms of the coexistence of urban and agricultural areas and biodiversity in a context of growing urbanisation. Within this plateau, we selected 12 ponds with surface areas ranging from 64 to 828 $m^2$ (mean ± SD: 540 ± 320 $m^2$; median: 566$m^2$) and with different potential exposures to agricultural and urban activities. The proportion of agricultural and urbanised surface around each pond was calculated in a 100 m radius buffer. As the aim was to study the effect on water quality of land use in the vicinity of ponds in a fairly fragmented landscape, we used a short-radius buffer zone. To calculate the different types of surface, we used the map of mainland France land cover produced by the Theia land data services using Sentinel 2 and Landsat 8 data (Theia 2017). The proportion of urbanised land was calculated by adding together three Theia categories: dense built-up areas, diffuse built-up areas and industrial and commercial areas. The proportion of agricultural land was calculated by adding two Theia categories: winter crops and summer crops. The remaining areas consist of forest and grassland. In the following, we refer arbitrarily to ponds A to L.

### 2.2 Sampling methods for invertebrates and for water and sediment

146 The sampling of invertebrates as well as water and sediment was systematically performed at
147 the same points in the 12 ponds. To ensure that water and sediment sampling and
148 invertebrate collection did not interfere with each other, they were carried out with a time
149 lag of about 1 week.

150

151 A preliminary analysis was conducted to determine the most appropriate timing for sampling
152 the invertebrates in the ponds. Several ponds were sampled every month from May to
153 October 2015 inclusive, which are the periods during which most species are active in our
154 geographical area. Eight of these ponds were included among the 12 ponds selected in our
155 study. Non-metric multi-dimensional scaling (NMDS) analysis of the collected taxa showed
156 that sampling in June and September covered almost all the diversity of the macroinvertebrate
157 species present in the ponds (Hanot, unpublished results). Sampling was thus performed in
158 June and September during 2016 and 2017 in the 12 ponds selected for the analysis. In the
159 following, we refer to the field campaigns as C1 (2016 June), C2 (2016 September), C3 (2017
160 June) and C4 (2017 September).

161

162 Macroinvertebrate sampling was carried out on two different main habitats on either side of
163 each pond to best represent the biodiversity of the ponds in a standardised manner. Sampling
164 was made using a pond-net with a 77 cm fiberglass handle, a trapezoid frame measuring 39 x
165 14 cm x 32 cm, a 1 x 1 mm mesh and a pocket depth of 40 cm. Samples were collected by
166 making an infinity symbol by hand eight times (i.e., a figure eight on its side), ending with a
167 fast move from the bottom to the top in the axis point of the symbol, to collect organisms
168 trapped in the vortex created by the sequence of movements. During this sequence, the frame

169 of the net pond was a few centimetres above the substratum, which allowed us to lift and
170 collect the benthos thanks to the upward current created by the “infinity” movement. To
171 sample each pond in the same way, the same operator made the movements at an arm’s
172 distance from the bank. In a small number of cases, the water level in the ponds was too low
173 to sample from the selected habitat. In this case, no collection was made at this point or at
174 any other to ensure that the samples were collected in the same location during the entire
175 study period.

176

177 During the collections in 2016, the content of the net pocket was quickly placed in a white
178 basin measuring 80 x 40 x 10 cm. All specimens were sorted by eye and fixed in absolute
179 ethanol 99% (Fisher Chemical, CAS 64-17-5) using entomological forceps. Collection stopped
180 when 5 min had elapsed without seeing any moving specimen. Thereafter, all samples were
181 kept at 7°C until the identification stage. During the 2017 campaigns, to speed up the sample
182 collection, water was squeezed from the net pocket as gently as possible to avoid destroying
183 the specimens, and the result was then fixed in absolute ethanol 99% in suitable containers
184 and kept at 7°C. Specimens were sorted using a Motic SMZ 171 binocular microscope and
185 entomological forceps, fixed again in absolute ethanol 99% and then kept at 7°C. Our sampling
186 protocol resulted in 88 invertebrate samples: 12 ponds x 2 points x 2 years x 2 seasons, minus
187 8 cases (4 ponds x 2 points) when the collection was not possible due to a low water level.
188 Invertebrate identifications (see next Section) were carried out separately for each sample.

189

190 In 2016 and 2017, in each of the 12 ponds, water and sediment samples were collected in
191 spring and autumn from the same two points selected for invertebrate sampling. The water

samples were taken approximately 1 m from the edge of the ponds using a stainless steel beaker with an extendable handle. Sediment sampling was performed with the same device used to collect the water samples.

**2.3 Identification of invertebrates**

Different books (Bameul 1985; Guignot 1947; Hansen 1987; Holmen 1997; Jansson 1986; Olmi 1976; Poisson 1957 ; Tachet et al. 2010) and websites (http://www.perla.developpement-durable.gouv.fr/, http://coleonet.de) were used as a reference to perform the morphological identification of all specimens under a Motic SMZ 171 stereo microscope. Identifications were made at the lowest possible taxonomic level. Specimens that could not be identified at the species level were identified at a higher taxonomic level, while adding a numerical suffix when more than one species was present (e.g., *Microvelia* sp.2). Because of these different levels of determination, we hereafter refer to specimens as “morphotaxa”, which are defined as taxa that share the same morphological characteristics. When it was not possible to link the different stages (larva, nymph, adult) to the same species, they were assigned to different morphotaxa. Some specimens of each taxon were kept in tubes of 2 mL, 5 mL or 40 mL according to their size, in absolute ethanol 99%, to be used as a reference. This made it possible to constitute a reference base for the invertebrates in the ponds by linking the reference specimens with their morphotaxon names. Each specimen of each sample was then identified using books, websites and the reference base. All specimens were counted by morphotaxa for each sample.

214 **2.4 Determination of water and sediment quality parameters, including trace elements**
215 **and organic pollutants**

216 The choice of contaminants assessed here was based on the local activities: cereals, maize,
217 rapeseed, sunflower, orchard and vegetable crops for pesticides; nearby roads for TE and PAH;
218 and the presence of humans, farms, and domestic pets for pharmaceuticals. Samples were
219 used to determine the main physicochemical parameters, including major and TE as well as
220 organic contaminants (PAH, pesticides and pharmaceuticals) as described in Nélieu et al.
221 (2020).

222

223 For the water samples, the following measurements were taken directly on site with probes:
224 pH, conductivity, temperature, dissolved oxygen (DO) and turbidity. Other data were obtained
225 rapidly in the laboratory (mainly within one day of sampling) using standardised methods:
226 dissolved organic carbon (DOC) by thermic oxidation and IR analysis of carbon dioxide,
227 chemical oxygen demand (COD, norm NF EN ISO 15705), suspended solids (SS, norm NF EN
228 872), nitrates ($NO_3^-$, norm NF EN ISO 10304-1), nitrites ($NO_2^-$, NF EN ISO 26777), total nitrogen
229 (TN) from the addition of Kjeldahl nitrogen (Kjeldahl method, norms NF EN 25663 and NF EN
230 ISO 11732) with nitrates and nitrites, total phosphorus (P, norm NF EN ISO 15681-2 and NF EN
231 ISO 6878.), anions (norm NF EN ISO 10304-1) and cations (norm NF EN ISO 14911), as well as
232 major and TE (norm NF EN ISO 17294-2: Al, As, B, Be, Cd, Cr, Cu, Fe, Hg, Mn, Ni, Pb, Sn, U and
233 Zn), 15 PAH, pesticides (25 herbicides, 1 safener, 7 fungicides and 2 insecticides) and 12
234 pharmaceuticals compatible with the multi-residue method applied after sample conservation
235 at -20°C (see the exhaustive list in Table S1). Details on the methods used for this
236 determination can be found in Nélieu et al. (2020).

237

238 Sediment samples were used to determine the contents in organic carbon (Corg), total
239 nitrogen (N) and thus C/N ratio (norms ISO 10694 and ISO 13878) as well as total major and
240 TE (after HF mineralisation and then ICP-AES or ICP-MS analysis according to the norm NF X
241 31-147/NF ISO 22036 - 17294-2), for the following: Cr, Cu, Ni, Zn, Co, Pb, Cd, Tl, Mo, Al, Ca, Fe,
242 K, Mg, Mn, Na, P ($P_2O_5$), Bi, In, Sb and Sn. All measurements were made in the Laboratory of
243 Soil Analysis of INRAe (Arras, France). The polycyclic aromatic hydrocarbons were analysed
244 according to the European standard NF EN 16181 (2018) by pressurised liquid extraction and
245 HPLC-fluorescence quantification. The same pesticides, metabolites and pharmaceuticals
246 selected for analysis in water (Nélieu et al., 2020) were also monitored in sediments (see
247 Appendix 1 for the analysis methods used for sediments).

248

249 Not all the measured pesticides and pharmaceuticals were detected in water and sediment
250 (no detection or values below the quantification limits). Therefore, our study is based on
251 fifteen herbicides (Atrazine, Atrazine-desethyl, Simazine, Terbuthylazine, Terbuthylazine-
252 desethyl, Clomazone, Diflufenican, Napropamid, Acetochlore, Alachlore, Dimethachlore,
253 Metolachlore, Chlorsulfuron, Metsulfuron-methyl and Nicosulfuron), seven fungicides
254 (Boscalid, Dimoxystrobine, Epoxiconazole, Hexaconazole, Metconazole, Picoxystrobine,
255 Tebuconazole), two insecticides (Imidacloprid and Pyrimicarb) and one pharmaceutical
256 (Carbamazepine).

257 All the data were added to the In.Do.Res repository (DOI not yet available).

258

259 **2.5 Statistical analysis**

Prior to the data analysis, the invertebrate samples collected from both points in each pond were pooled. This allowed us to produce a contingency table containing the number of each morphotaxon for each pond and each sampling date. For the water quality parameters, TE and organic pollutants, we computed the mean of the two sample point values. Although the surface area of ponds is important, as pointed out in the introduction, it could not be included as a factor in the analyses because we only have a single measurement. All statistical analyses were performed with R 3.6.1 (Team 2020).

We performed analyses based on morphotaxa abundance and presence-absence. However, due to differences in specimen determination levels, the analyses may be biased and should therefore be taken with caution. To explore invertebrate community diversity, we computed the morphotaxa richness (alpha diversity), Shannon diversity index and Pielou evenness for each pond and each field campaign with the specnumber function in vegan 2.5-6 (Oksanen et al. 2019). We tested the effect of individual ponds and field campaign on these parameters with an analysis of variance with additive effects of pond and field session followed by a pairwise comparison with Tukey's HSD test.

To study the dissimilarities between invertebrate communities, we computed the total beta diversity within each field campaign across all ponds using the beta.multi function of the betapart package 1.5.2 (Baselga et al. 2020). This function partitions the total beta diversity into two additive components, turnover and nestedness, which reflect species replacement and species richness difference, respectively (Baselga et al. 2020).

282 We calculated the contribution of each pond to beta diversity, that is, the local contribution
283 to beta diversity (hereafter LCBD) following Legendre and Cáceres (2013) using the beta.div
284 function of the adespatial package 0.3-8 (Dray et al. 2019). As pond dissimilarity may be
285 different if computed with the abundance or presence-absence (PA) of morphotaxa, we
286 calculated both using the Hellinger and Jaccard dissimilarity coefficients, respectively. The
287 LCBD values, which represent the uniqueness of a pond in terms of taxa composition, were
288 tested for significance with the null hypothesis of a random distribution of species among
289 ponds within a sampling campaign (Legendre and De Cáceres 2013). We also computed the
290 species contributions to beta diversity (hereafter SCBD) to identify the morphotaxa that
291 contribute the most to beta diversity. This last index is calculated only with morphotaxa
292 abundance. We explored the temporal effects (year and season) on morphotaxa assemblages
293 but as the results are not robust, we present them in Appendix 2.

294

295 To explore the relationship between the parameters describing diversity and land use, we
296 tested the effects of the proportion of urban, forested and agricultural areas on morphotaxa
297 richness, diversity, equitability and LCBD with a linear mixed effect model with the field
298 campaign as a random effect. We used the lme4 package, v.1.1-23 (Capps et al. 2015).

299

300 Finally, we explored the relationship between the environmental parameters and the
301 macroinvertebrate assemblages to identify the parameters that best explain the community
302 structures. The concentrations of water TE, PAH and pharmaceuticals were very low, being at
303 the detection limit; for this reason, we did not include them in the analyses. Prior to the
304 analyses, to reduce the high number of TE in the sediments, we performed principal

305 component analysis (PCA) in all ponds for the four field campaigns. The first two axes of the
306 PCA explain 68.67% of the variability (Figure S1). The ponds are arranged on the first axis (TE1)
307 from low to high TE concentrations. The second axis (TE2) discriminates ponds with high
308 concentrations of TE (Sb, Cd) from those with high concentrations of major elements (Na, Mg,
309 Fe). We collected the coordinates of each pond-field campaign combination for the first two
310 axes of the PCA and then used them in the following statistical analyses as summaries of each
311 contaminant group effects on ponds. We summed the concentration of the different PAH and
312 used the total PAH concentration in the following analyses. We used redundancy analysis
313 (RDA) (Borcard et al. 2011) and ran two analyses, one with the Hellinger-transformed
314 abundances of morphotaxa and another with the PA of morphotaxa. In the RDA, the response
315 matrix is the abundance or PA of morphotaxa in all ponds and the four field campaigns,
316 whereas the explanatory matrix is the environmental parameters, including the contaminants
317 for the same pond-field campaign combination. We used the following parameters in the RDA:
318 (1) for water: conductivity, suspended matter, COD, DOC, concentrations of TN and
319 phosphorus, orthophosphate, ammonium, organic carbon, herbicides and insecticides; and
320 (2) for sediments: herbicides, fungicides, insecticides, pharmaceuticals, PAH and the first two
321 PCA axes performed with TE.
322
323 We used the rda function of the vegan package v2.5-6 (Oksanen et al. 2019). We tested the
324 significance of the two RDA results by the permutation of the overall analysis and each axis.
325 The two RDA were significant with a threshold level of 5%. We tested for linear dependencies
326 among the explanatory variables and computed the variance inflation factors (VIF) of the
327 variables with the vif.cca function. We computed the adjusted $R^2$ with the RsquareAdj
328 function. The VIF was very high for several explanatory variables and so to reduce the

correlations between them, we computed a forward selection of the explanatory variables using the forward.sel function. The method produces parsimonious models, which we have tested by permutation and VIF.

334 - **Results**

335 **3.1 Morphotaxa richness, Shannon diversity index and evenness**

336 In total, we identified 236 morphotaxa, which represent a total of 22 orders and 54 families
337 including 13 orders and 42 families for arthropods and 8 orders and 37 families for insects
338 (some morphotaxa could not be assigned to an order or family). The Baetidae *C. dipterum* was
339 ubiquitous during campaigns C1, C2 and C3 but totally absent in C4. Morphotaxa richness
340 ranged from 7 to 49 with a median value of 25. Statistical analyses show a weak effect of pond
341 on morphotaxa richness (Figure 1 and Table S2). Morphotaxa richness was significantly higher
342 in pond J than in ponds E ($p=0.023$) and I ($p=0.035$). Morphotaxa richness changed between
343 field campaigns ($p\leq 0.001$, Table S1), being significantly higher in C1 and C2 than in C4
344 (respectively $p=0.004$ and $p=0.008$). The Shannon index and evenness ranged from 1.07 to 2.9
345 and from 0.33 to 0.82, respectively. The field campaigns and ponds had no effect on the
346 Shannon index. Only the field campaign had an effect on evenness ($p=0.02$) with a higher
347 evenness in C4 than in C1 ($p=0.02$).

348

349 **3.2 Beta diversity: Spatial dissimilarities between ponds**

350 The total beta diversity based on morphotaxa PA was very similar for each field campaign with
351 values between 0.92 and 0.93 (Table S3; for comparison, results for analyses based on
352 morphotaxa abundances are in Table S2). The turnover, which reflects the level of species
353 replacement between ponds as opposed to species loss, represents between 88% and 90% of
354 this total beta diversity. The LCBD of each pond based on morphotaxa PA varied across the
355 field campaigns (Figure 2, Table S3). Pond E has a significant contribution to regional
356 biodiversity in C1 and C2. Pond I makes a significant to highly significant contribution during
357 the four field sessions and ponds J and L in C4 and C3, respectively.

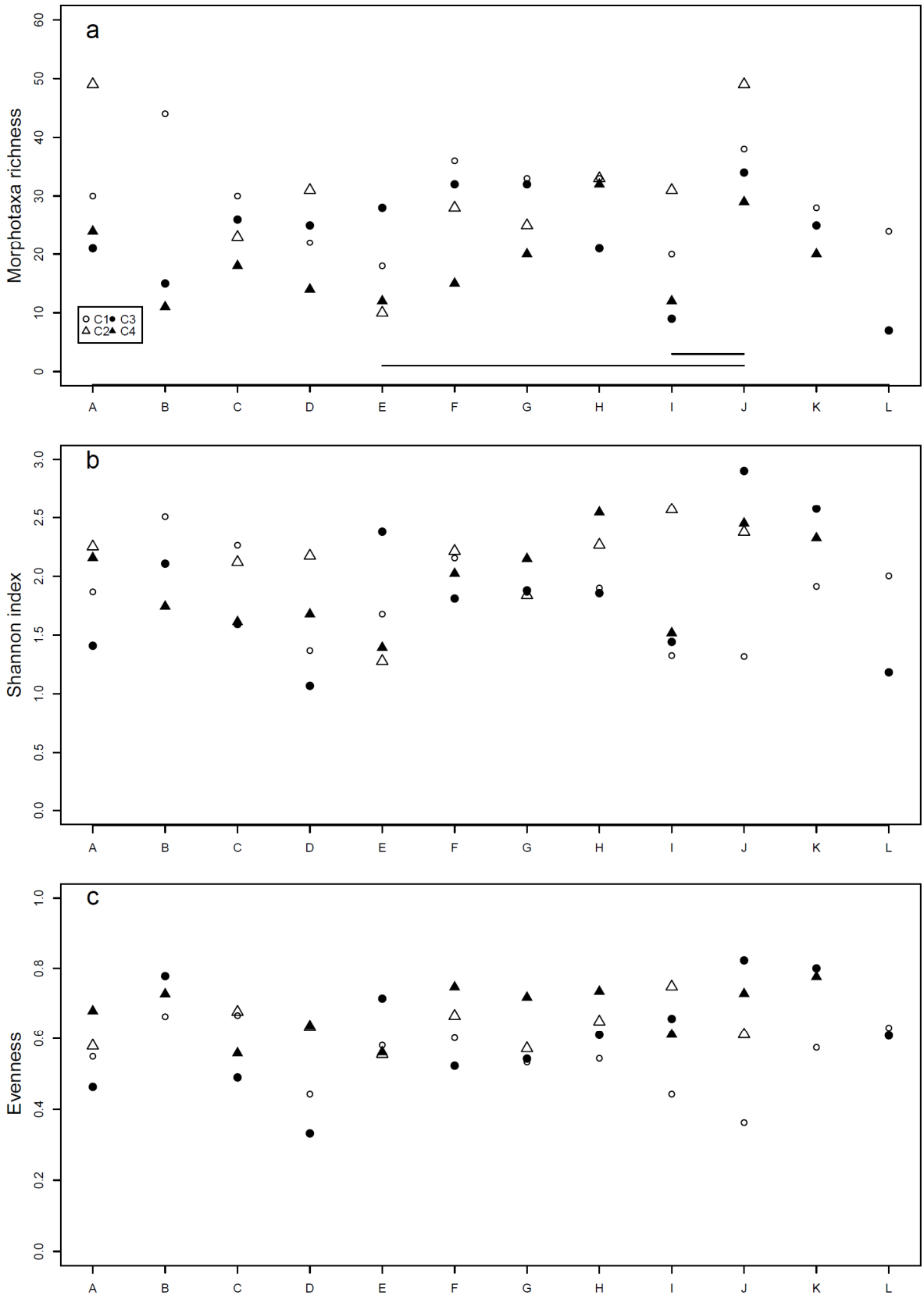


**Figure 1. Morphotaxa richness (a), Shannon index (b) and evenness (c) in the 12 ponds for the four field sessions.** The legend for the four field sessions is given in panel (a). The horizontal segments in panel (a) link the ponds significantly different.

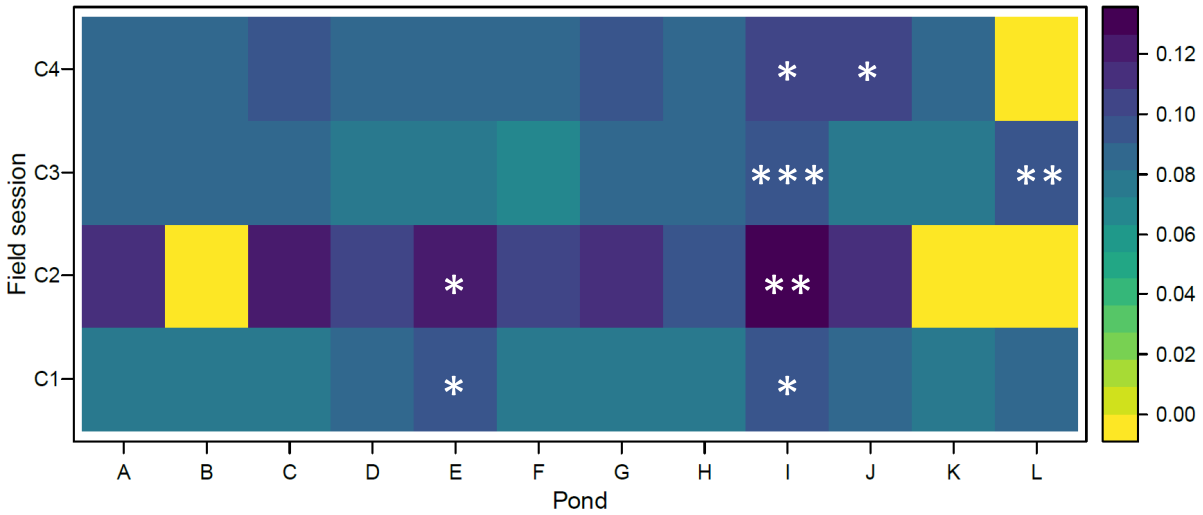


**Figure 2. Local contribution to beta diversity (LCBD) of each pond for the four field sessions.** The LCBD is computed with the presence-absence of morphotaxa. The LCBD are computed independently for each field session. The symbols show the significant contributions with: ***: $p \leq 0.001$; **: $0.001<p \leq 0.01$; *: $0.01 < p \leq 0.05$.

To shed light on the LCBD results, we briefly present the Species (here morphotaxa) Contribution to Local Biodiversity (SCBD) based on morphotaxa abundances. The analysis reveals that a relatively low number of morphotaxa explains most of the dissimilarities among ponds. The 15 morphotaxa with the highest contributions are listed in Table S5 and the comprehensive list of morphotaxa SCBD is provided in Table S6. The 15 morphotaxa with the highest contributions account for 83%, 73%, 71% and 70% of the total SCBD in the C1, C2, C3 and C4 field campaigns, respectively. Some are common to all four campaigns (*Asellus* sp., two different *Chaoborus* sp., different Chironomini morphotaxa) or to three campaigns (*Cloeon dipterum*; Clitellata; *Physella acuta*; *Valvata macrostoma*).

380 **Table 1. Results of the redundancy analyses (RA) parsimonious models**

| Variables | Cumulative adjusted $R^2$ | F value | *p* value |
|---|---|---|---|
| Water conductivity | 0.032 | 2.401 | 0.001 |
| Sediment pharmaceuticals | 0.057 | 2.111 | 0.001 |
| Water ammonium concentration | 0.077 | 1.922 | 0.006 |

381

382

383 **3.3 Relationship between land use, environmental parameters and macroinvertebrate**

384 **assemblages**

385 The linear mixed effect model testing the effects of the different types of land use around the
386 ponds on parameters characterising the macroinvertebrate diversity showed that equitability
387 increased significantly with the proportion of agricultural area in pond vicinity (p=0.026).
388 Morphotaxa richness and diversity, and LCBD were not significantly affected by land use.
389 To clarify the relationship between the environmental parameters for water and sediment
390 and the macroinvertebrate assemblages, we ran redundancy analysis (RDA) with morphotaxa
391 PA. The initial model with all the environmental parameters was significant (p = 0.009). As
392 some of these parameters had strong collinearities (as shown by the high VIF values), a
393 forward model selection procedure was used to obtain a more parsimonious model. The final
394 model describing the morphotaxa PA contained three explanatory variables: conductivity,
395 pharmaceutical concentration in sediments and ammonium concentration in water (Table 1).
396 The parsimonious model is highly significant (Df = 3, 40, F-value = 1.740, p=0.002) with no
397 strong collinearity between the variables (all VIF are around 1) and is reduced to one
398 significant canonical axis (Figure 3). Pond C is associated with high concentration of
399 pharmaceuticals in the sediment; pond G is also associated with high concentration of
400 pharmaceuticals in the sediment in addition to high water conductivity.

401

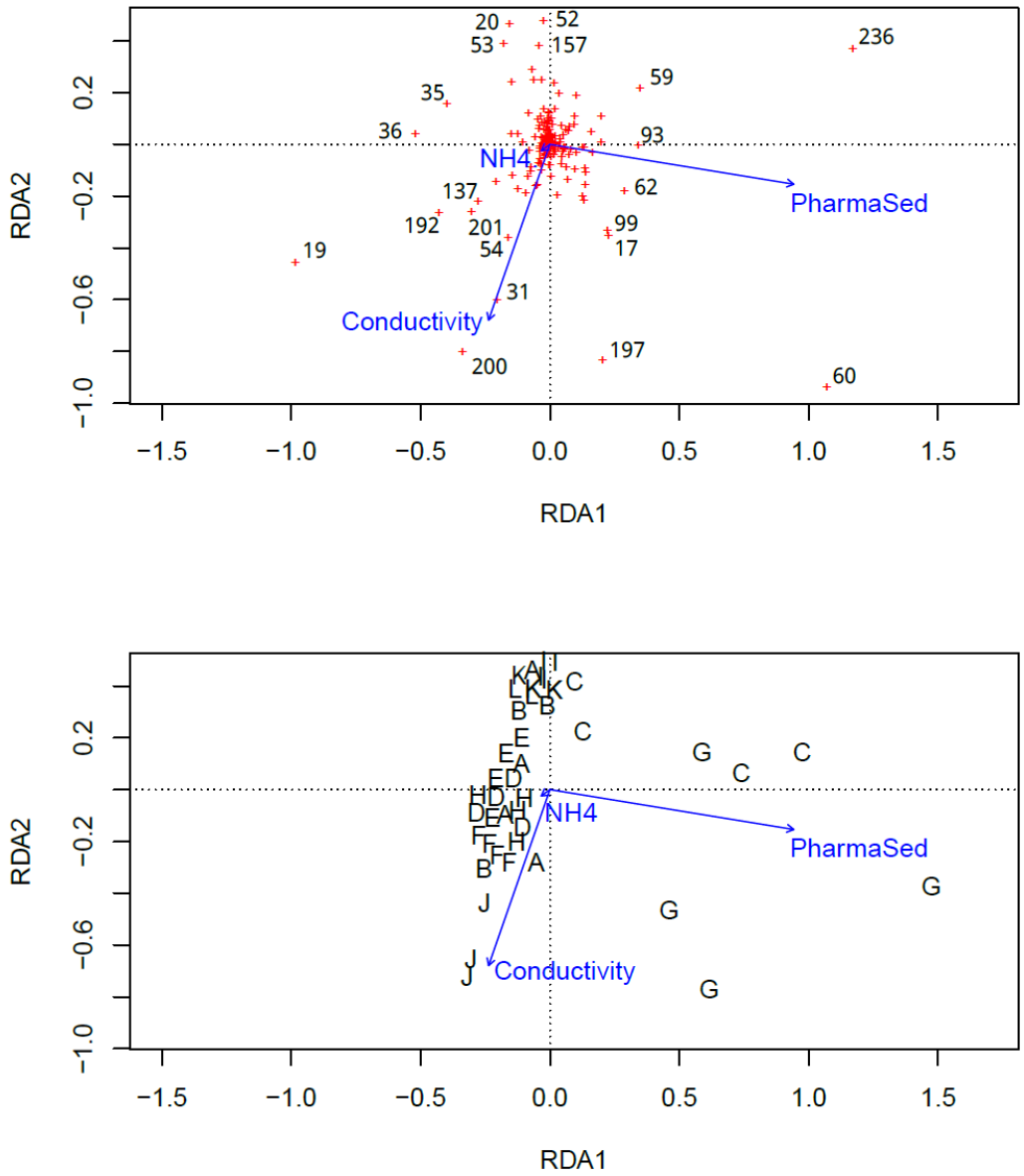


**Figure 3. Parsimonious redundancy analysis (RDA) based on morphotaxa presence-absence.** The biplots show the variables (blue arrows) with either the morphotaxa (numbers, panel a) or the ponds (letters, panel b). The number of morphotaxa is shown in brackets: Anophelinae 01 (17); *Asellus* sp. (19); Baetidae 02 (20); Ceratopogoninae 02 (31); *Chaoborus* sp. 01 (35); *Chaoborus* sp. 02 (36); Chironomini 04 (52); *Chironomus* sp. 01 (53); *Chironomus* sp. 02 (54); Clitellata 01 (59); *Cloeon dipterum* (60); *Coenagrion* sp.01 (62); *Dugesia* sp. 2 (93); *Erythromma viridulum* (99); Hesperocorixa 03 (137); *Hygrotus inaequalis* (157); *Physella acuta* (192); *Plea minutissima* (197); *Potamopyrgus antipodarum* (200); *Proasellus* sp. (201); and *Valvata macrostoma* (236). Conductivity: water conductivity; PharmaSed: concentration of pharmaceuticals in sediment; NH4: water ammonium concentration.

Pond J is associated with high water conductivity. Ponds, A, B, D, E, F, H, I, K and L are less affected by environmental parameters although ponds K and L seem associated with low

water conductivity. A few morphotaxa stand out and are associated with some environmental variables (morphotaxa scores in the RDA are in Table S7). *Valvata macrostoma* is associated with high pharmaceutical concentrations in sediment; *Potamopyrgus antipodarum* is associated to high water conductivity; *Cloeon dipterum* and *Plea minutissima* are associated to high values both variables. Baetidae 02, *Chironomus* sp. 01, *Chironomus* sp. 02 and *Hygrotus inaequalis* are associated with ponds in which these environmental parameters have low values.

## Discussion

### 4.1 Morphotaxa distribution in the ponds and effects of land use

We changed our experimental design for macroinvertebrate sampling between the 2 years of the study. The results show a greater morphotaxa richness in the first year compared with autumn in the second year. The morphotaxa diversity was not affected and the eveness slightly affected by the field campaign. As a consequence, it is difficult to conclude on the effects of the protocol change, as the effects may be small or may have been buffered by an annual effect.

Our analysis is based on morphotaxa determined at different taxonomic levels, which is open to criticism. The aim of our work is to compare the response of assemblages to the presence of pollutants and not to compare the diversity of the ponds studied with other ponds. In this sense, questions of determination level are less important, since the same precision has been maintained for all samples. Furthermore, studies on interaction networks have shown that the level of determination of specimens has little effect on network characteristics, provided that this level of determination does not fall below too high a threshold (Llopis-Belenguer et al. 2023; Renaud et al. 2020).

Among the 236 macroinvertebrate morphotaxa identified, we did not find endangered, vulnerable or even rare species. However, we observed exotic species: the molluscs *Potamopyrgus antipodarum* and *Physella acuta* and the crayfish *Procambarus clarkii* listed as an invasive alien species in the European Union (European Union 2016). Molluscs and crustaceans are the most frequent freshwater macroinvertebrate invaders (Oertli and Parris 2019; Patoka et al. 2017). The pet trade is one of the main introduction pathways, and both mollusc species can "hitchhike" on intended shipments (Patoka et al. 2017). One individual *P. clarkii* was found in pond F in C3. Pond F also hosted the two exotic molluscs, and pond J hosted *P. antipodarum*. We should stress here that pond F, though in a forested area, is located on the Paris-Saclay university campus with heavily frequented paths in close vicinity. The campus is open to the public, which may favour the dissemination of invasive species.

Overall, our results show that the ponds distributed along an urbanisation gradient are quite dissimilar, as beta diversity relies mostly on morphotaxa turnover with a comparable morphotaxa diversity. No pond stands out consistently across the four sampling campaigns in terms of the morphotaxa contribution to regional diversity except for pond I and, to a lesser extent, pond E. The LCBD indicates the uniqueness of communities either because they are rich and host typical taxa or because they are degraded with a limited number of common taxa (Legendre 2014; Legendre and De Cáceres 2013). Here, the uniqueness highlighted by the significant LCBD points to degraded ponds. For instance, pond I has a low diversity, and its most striking feature is the absence of Baetidae in C4, whereas the mean abundance in other ponds was 81.2 individuals (± 47.0 SD). Despite the restrictions described above, we have calculated the SCBD, based on morphotaxa abundances, because it supports the idea that some ponds stand out because they are degraded. The lists of morphotaxa contributing the

464 most to regional diversity encompass common morphotaxa, some of which are characteristic
465 of degraded communities such as Chironomidae, *Chaoborus* sp., *C. dipterum*, and so on. For
466 instance, in C2, pond C was characterised by high abundances of *Chaoborus* sp. 01, *C.*
467 *dipterum*, *Corixa* sp. and *V. macrostoma*, and pond E by *C. dipterum*, Orthocladiinae and
468 Tanytarsini 01. Pond L in C3 had a low diversity and was dominated by *Notonecta* sp. 01. The
469 uniqueness of a community as shown by a high LCBD may also indicate the presence of
470 invasive species (Legendre 2014). However, the two ponds hosting exotic species as well as an
471 invasive species had no significant LCBD values.
472
473 We initially hypothesised that local macroinvertebrate diversity is higher in ponds located in
474 rural areas than in those located in agricultural or urban areas. This hypothesis is supported
475 by different studies (Blicharska et al. 2017; Johnson et al. 2013; Noble and Hassall 2015;
476 Thornhill et al. 2017). However, our results do not support this hypothesis, as we found only
477 an effect of the proportion of agricultural area on morphotaxa evenness. Our results likewise
478 do not support the second hypothesis regarding rare and pollutant-sensitive morphotaxa, as
479 we do not observe any of them. On the contrary, we found invasive and exotic species. LCBD
480 values could help to identify these ponds (Legendre 2014), although they only identified pond
481 J, probably because of the very high density of *P. antipodarum*.
482
483 **4.2 Effects of environmental parameters on macroinvertebrate assemblages**
484 Our results showed that among the numerous environmental parameters and pollutants
485 measured in water and sediment, very few are critical for macroinvertebrate assemblages in
486 ponds. The concentration of pharmaceuticals in sediment and water conductivity are the most
487 structuring parameters of macroinvertebrate assemblages. Although these results should be

488 treated with caution, the analysis with morphotaxa abundances reveal other important
489 parameters as the concentration of fungicides in sediment as well as MTE1, insecticides,
490 organic carbon, and COD in water. Nélieu et al. (2020) highlighted the high environmental risks
491 due to water column pesticide concentrations in several ponds. Pesticides other than
492 insecticides do not seem to be critical factors to explain the macroinvertebrate assemblages
493 observed here. In contrast to sediment pollution, pollutants measured in the water column
494 provide a snapshot into water quality; sediment pollution is relatively stable over time, and
495 the measurements are more reliable as an indicator of pollution level (Casey et al. 2007; Sun
496 et al. 2019). Conductivity is a general indicator of the presence of many ions in the solution,
497 which is consistent with urban pollution associated with de-icing salts and TE (Brand et al.
498 2010; Oertli and Parris 2019; Wu et al. 2020). Surprisingly, conductivity is not associated with
499 TE concentrations in our study, suggesting that these two factors do not filter morphotaxa in
500 the same way in different ponds. Conductivity is associated with forest pond J and, to a lesser
501 extent pond G,in addition to Ceratopogoninae, *P. antipodarum* and *P. minutissima*. Though
502 in a rural area, pond J is bordered by a road, which may explain the high conductivity.
503
504 Based on morphotaxa abundances, we found two groups of parameters, COD and TE1, in
505 water on the one hand, and fungicide and pharmaceuticals in sediment with dissolved
506 insecticides on the other, generally in the same ponds but at different sampling campaigns.
507 These ponds include G, C, B and E. Ponds B, C and E are located in an agricultural area, ponds
508 C and G are near a farm and pond G is near a medical center. The Chironomidae
509 Orthocladiinae, the mayfly *C. dipterum*, the mollusc *V. macrostoma* and the annelid *Clitellata*
510 sp. characterise the assemblages found in these ponds. These morphotaxa, in particular

511 Chironomidae and annelids, are typical of aquatic systems embedded in a degraded
512 environment (Hill and Wood 2014; Mackintosh et al. 2015; Wood et al. 2001)
513
514 When considering the PA of morphotaxa, water conductivity is associated with pond J
515 described above.
516
517 Compared with the analysis based on abundances, ponds C and G are characterised by high
518 concentrations of pollutants (i.e., pharmaceuticals and fungicides in the sediment). Another
519 set of ponds is associated with high water conductivity and major ion concentrations. This set
520 includes pond J and, to a lesser extent, ponds A, K and L, characterised by the presence of the
521 dipterans Ceratopogoninae, *Chaoborus* sp. and *Chironomus* sp., the Heteroptera *P.*
522 *minutissima* and the Crustacea *Proasellus* sp. These ponds are mostly surrounded by
523 grasslands and forest, with pond K being the most urbanised pond with nearby dwellings. This
524 points to diffuse pollution associated with road traffic and occasional human activities. In both
525 analyses, the ubiquitous *C. dipterum* is distinguished by its association with high dissolved ion
526 concentrations and, to a lesser extent, by pharmaceuticals in sediment.
527
528 Legendre (2014) recommends using PA dissimilarity coefficients when community
529 assemblages are characterised by a high turnover and quantitative indices when the
530 assemblages differ in terms of abundances rather than species diversity. In our study, the high
531 turnover in macroinvertebrate assemblages favours an analysis based on PA, thus concluding
532 the RDA with the least factors: pharmaceuticals sediment and conductivity and ammonium
533 concentration in water. Our third hypothesis is thus partially validated, as contaminants allow
534 us to discriminate several ponds characterised by certain morphotaxa.

535

536 **4.3 Characteristics of the peri-urban environment**

537 The absence of a clear relationship between land use and morphotaxa diversity suggests that
538 the presence of roads, buildings, or impervious surfaces in close vicinity to the ponds is not a
539 critical parameter to explain the observed patterns of morphotaxa diversity at the regional
540 scale. Instead, traces of particular activities influence morphotaxa diversity. For instance, pond
541 A is located in a forest, which explains the hunting cartridges found in and around the pond.
542 We did not find a high level of contaminants in this pond, although its conductivity may be
543 due to the cartridges. Situated between a forest and fields, pond L is not far from residential
544 buildings; people walk to this pond and brush their dogs there (we found a bristle of hairs),
545 also allowing them to swim in the water. In this pond, Nélieu et al. (2020) found high
546 imidacloprid concentrations, which is a veterinary pharmaceutical used to treat dog fleas and
547 ticks. Ponds B and C are both located near farms. We observed that the farmer washed his
548 tractor and equipment in one pond, with the wastewater running off into the pond. Pond G is
549 located near a farm and medical center where carbamazepine, a human anti-epileptic, is used.
550 Pharmaceuticals are markers of human activities, and carbamazepine, which is resistant to
551 biodegradation, is only used by humans (Kasprzyk-Hordern et al. 2009). These observations
552 illustrate the multi-functionality of peri-urban areas with a mixture of different users
553 (Friedmann 2016; Zoomers et al. 2017) at the local scale. Peri-urban ponds combine
554 contaminants typical of rural and urban environments, that is, a runoff of excess nitrogen (and
555 phosphorus) and an influx of heavy metals and salt from road applications (Wanek et al. 2021).
556 To this list, we may add pesticides and pharmaceuticals.

557

558 In their review, Oertli and Parris (2019) showed the diversity of criteria used to quantify the
559 urbanisation of a site (presence of buildings, roads, etc.). We suggest here that local and
560 recurrent human actions may blur these categories, particularly in semi-urban areas. Though
561 embedded in a semi-natural matrix, peri-urban ponds are easily reached and may encounter
562 small, chronic and various perturbations. Several authors have advocated for well-managed
563 urban ponds to provide high-quality habitats and support greater biodiversity (Oertli and
564 Parris 2019; Perron et al. 2021). Peri-urban areas tend to be well described, and their potential
565 contribution to sustainable development thus becomes more evident (Wandl and Magoni
566 2017). As the water quality of peri-urban ponds tends to be more similar to urban ponds than
567 rural ones (Wanek et al. 2021), efforts should be made to manage these systems, especially
568 since they connect rural and urban systems in the blue grid. The use of the LCBD is an
569 interesting approach to identify the ponds to restore (Legendre 2014).
570

571 ## 5. Conclusion

572 Our study of macroinvertebrates and water and sediment contaminants in 12 peri-urban
573 ponds over 2 consecutive years reveals a high morphotaxa turnover with the absence of rare
574 and pollutant-sensitive morphotaxa. The macroinvertebrate assemblages were relatively
575 stable, and those contributing the most to regional biodiversity are typical of degraded ponds.
576 The pollutants best describing macroinvertebrate PA in assemblages are pharmaceuticals in
577 sediment and conductivity and ammonium concentration in water. Although an
578 environmental risk due to water column pesticides could be estimated, this factor is not
579 structuring for macroinvertebrate community. Peri-urban areas are characterised by multi-
580 functionality with a mixture of different uses and users. Ponds located in these environments

are exposed to various human activities, leading to small, chronic and diverse contaminations that affect macroinvertebrate abundance and community structure.

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**Data, scripts, code, and supplementary information availability**

All the data and scripts are available online: https://doi.org/10.48579/PRO/TX0PU9.
Supplementary information is available online: https://doi.org/10.48579/PRO/TX0PU9.
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**Conflict of interest disclosure**

The authors declare that they comply with the PCI rule of having no financial conflicts of interest in relation to the content of the article. The authors declare the following non-financial conflict of interest: Isabelle Lamy is a recommender of PCI Ecotoxicology.

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