



HAL
open science

Combined Chemical and Ecotoxicological Measurements for River Sediment Management in an On-Land Deposit Scenario

Jean-Philippe Bedell, André-Marie Dendievel, Antoine Gosset, Brice Mourier

► **To cite this version:**

Jean-Philippe Bedell, André-Marie Dendievel, Antoine Gosset, Brice Mourier. Combined Chemical and Ecotoxicological Measurements for River Sediment Management in an On-Land Deposit Scenario. Archives of Environmental Contamination and Toxicology, inPress, 10.1007/s00244-023-00997-0 . hal-04107957

HAL Id: hal-04107957

<https://hal.science/hal-04107957>

Submitted on 27 Jul 2023

HAL is a multi-disciplinary open access archive for the deposit and dissemination of scientific research documents, whether they are published or not. The documents may come from teaching and research institutions in France or abroad, or from public or private research centers.

L'archive ouverte pluridisciplinaire **HAL**, est destinée au dépôt et à la diffusion de documents scientifiques de niveau recherche, publiés ou non, émanant des établissements d'enseignement et de recherche français ou étrangers, des laboratoires publics ou privés.

1 **Combined chemical and ecotoxicological measurements for river sediment management in**
2 **an on-land deposit scenario**

3 Bedell Jean-Philippe*, Dendievel André-Marie, Gosset Antoine, Mourier Brice

Univ Lyon, Université Claude Bernard Lyon 1, CNRS, ENTPE, UMR 5023 LEHNA, F-69518 Vaulx-en-Velin,
France

4 * *Corresponding author: bedell@entpe.fr*

5

6

7

8

9 ABSTRACT

10 Sediment management along engineered river systems includes dredging operations and sediment deposition in the
11 sea (capping) or on land. Thus, determining the ecotoxicological risk gradient associated with river sediments is
12 critical. In this study, we investigated sediment samples along the Rhône River (France) and conducted
13 Environmental Risk Assessment (ERA) tests with the idea to evaluate them in the future for deposit on soil. Based
14 on an on-land deposit scenario, the capacity of the sediment samples from four sites (LDB, BER, GEC, and TRS) to
15 support vegetation was evaluated by characterising the physical and chemical parameters (pH, conductivity, total
16 organic carbon, grain size, C/N, potassium, nitrogen, and selected pollutants, including polychlorinated biphenyls
17 (PCBs) and metal trace elements. All tested sediments were contaminated by metallic elements and PCBs as follows:
18 LDB > GEC > TRS > BER, but only LDB had levels higher than the French regulatory threshold S1. Sediment
19 ecotoxicity was then assessed using acute (plant germination and earthworm avoidance) and chronic (ostracod test
20 and earthworm reproduction) bioassays. Two of the tested plant species, *Lolium perenne* (ray grass) and *Cucurbita*
21 *pepo* (zucchini), were highly sensitive to sediment phytotoxicity. Acute tests also showed significant inhibition of
22 germination and root growth, with avoidance by *Eisenia fetida* at the least contaminated sites (TRS and BER).
23 Chronic bioassays revealed that LDB and TRS sediment were significantly toxic to *E. fetida* and *Heterocypris*
24 *incongruens* (Ostracoda), and GEC sediment was toxic for the latter organism. In this on-land and spatialised
25 deposit scenario, river sediment from the LDB site (Lake Bourget marina) presented the highest potential toxicity
26 and required the greatest attention. However, low contamination levels can also lead to potential toxicity (as
27 demonstrated for GEC and TRS site), underlining the importance of a multiple test approach for this scenario.

28 KEY WORDS:

29 *Cucurbita pepo*; Dredged sediments; Environmental Risk Assessment (ERA); *Eisenia fetida*; *Heterocypris*
30 *incongruens*; *Lolium perenne*; Metals; PCBs; Rhône River;

31

32 **Introduction**

33 Sediment can be both a sink and source of contaminants for the overlying water column and biota,
34 especially when dredged from a river (Perrodin et al., 2006; Volatier et al., 2009; Bedaa et al., 2020; Heise et al.,
35 2020). Several management options can be considered after dredging river sediments, such as capping, on-land
36 deposits, or discharge into water by resuspension. In the context of the European Water Framework Directive
37 (2000/60/EC) and, more recently, the Waste Framework Directive (2008/98/EG), the reuse of dredged sediments
38 requires a demonstration of environmental acceptability based on the assessment of physical and chemical
39 characteristics, especially by applying HP14 and NF EN 12920+A1 methodologies (Lecomte et al., 2019). In the
40 case of an on-land sediment deposit, an environmental risk assessment (ERA) is usually performed to assess the risks
41 to organisms and ecosystems surrounding the sediment deposit location (e.g. watercourses, wetlands, terrestrial
42 ecosystems) and groundwater quality (Perrodin et al., 2006; Pesce et al., 2020). Therefore, assessing the contaminant
43 content and potential toxicity of sediments to organisms growing on the surface or living in such deposits is
44 necessary. The sediment matrix approach is generally used to test seed germination, plant growth, earthworm
45 avoidance, and reproduction. Although it does not enable the determination of the substance(s) responsible for the
46 observed effect, it has the advantage of characterising hazards and exposure according to the contaminant load.

47 In risk evaluation, the first step is to calculate a contamination index based on a comparison of contaminant
48 concentrations with quality thresholds from the literature (e.g. PEC, probable effect concentration threshold) before
49 determining a risk quotient (MacDonald et al., 2000a,b). The risk quotient (RQ) was used to assess the effect of
50 mixtures of contaminants and compare samples with various compounds (MacDonald et al., 2000a). The RQ can be
51 defined as the ratio between the measured concentration of each pollutant in the environment (dredged sediments)
52 and the corresponding predicted no effect concentration for sediment-dwelling organisms ($PNEC_{sed}$) (ECB, 2003).
53 The use of $PNEC_{sed}$ is valuable for considering a wide range of micropollutants that are not included in the PEC
54 (MacDonald et al., 2000a). Then, all RQ are added to the Risk Quotient mixture (RQ_{mix}) to estimate the global risk
55 based on a concentration addition model (Backhaus et al., 2012). As an example, the RQ_{mix} approach was recently
56 used to assess the risk from the effects of a cocktail of contaminants in dated sediment from backwater sites along
57 the Rhône, highlighting a high risk of mixture dominated by metals before 2005, and the time span since 2005 has
58 proven to be mainly driven by POPs (Liber et al., 2019; Dendievel et al. 2020b).

59 Another approach to qualify the risk is to implement bioassays to evaluate the effects of exposure on fauna or plants.
60 Ecotoxicological tools can provide added value for monitoring the true state of river quality, considering
61 contaminants potentially present at concentrations high enough to cause biological effects (Martinez-Haro et al.,
62 2022). Bioassays performed on sediments dredged from canals have previously shown the importance of
63 phytotoxicity tests in determining the risks related to their deposits (Bedell et al., 2003; Perrodin et al., 2006;
64 Lecomte et al., 2019). Studies on seed germination have demonstrated a significant reduction in the growth of some
65 plant species in contact with metal-contaminated soils and sediments (Adam and Duncan, 2002; Chen et al., 2002;
66 Bedell et al., 2003; Czerniawska-Kusza et al., 2006). Seed germination studies have also been used to evaluate the
67 toxicity of anthropogenic matrices such as manure and wastewater treatment plant sludge (Fuentes et al., 2004;

68 Oleszczuk, 2008; Czerniawska-Kusza and Kusza, 2011; Oleszczuk et al., 2012). Acute ecotoxicity tests based on
69 earthworm avoidance bioassays (ISO 11268-1, 1993) and chronic toxicity tests are usually combined to determine
70 their effects on reproduction (ISO 11268-2, 1998) and to assess soil toxicity (Heupel, 2002; Greenslade and
71 Vaughan, 2003; Da Luz et al., 2004, Lecomte et al., 2019). The avoidance test can be used to distinguish soils or
72 matrices subjected to different anthropogenic stresses (Sousa et al., 2008; Alvarenga et al., 2012) or in various
73 environmental conditions (Garcia et al., 2008; Buch et al., 2013). Earthworm reproduction tests have been used to
74 assess chlorpyrifos-contaminated soils (Zhou et al., 2007), petroleum hydrocarbons, lead-contaminated soils
75 (Whitfield Aslund et al., 2013; Luo et al., 2014), urban soils (Hankard et al., 2005) and other anthroposols (Coehlo et
76 al., 2018). Another frequently used bioassay is the standardised ostracod toxicity test (using *Heterocypris*
77 *incongruens*). Indeed, another study, performed on 33 types of sediments, highlighted the potential of this test to
78 provide a reliable and sensitive alternative for sediment toxicity assessment (Belgis et al., 2003). This ostracod test
79 was also implemented to study the effects of contaminants on ecological functions supported by sediment
80 communities and showed high toxicity of Cu-spiked and Cu-plus-As-spiked sediments and low toxicity of As-spiked
81 sediments (Pesce et al., 2020). This test has also been used successfully in wetland and river contexts for the
82 treatment of landfill leachate (Buitrago et al., 2013), urban sediment toxicity assessment (Gonzalez-Merchan et al.,
83 2014), and assessment of hospital effluents (Mubedi et al., 2013; Wang et al., 2009; Perrodin et al., 2013).

84 Because chemical analyses alone do not necessarily reflect the bioavailability and toxic action of measured
85 contaminants, multidisciplinary approaches are required to assess the chemical, biological, and toxicological impact
86 of complex mixtures of contaminants (Todaro et al., 2019). In this respect, ecotoxicological bioassays are essential
87 complementary tools for investigating the link between contaminant contents and ecological responses and are even
88 more powerful in detecting the effects of pollutant mixtures that are not necessarily targeted (Heise et al., 2020).
89 However, to our knowledge, only a few studies have combined chemical and ecotoxicological approaches to assess
90 the toxicity of sediments in on-land deposit scenarios. Todaro et al. (2019) highlighted an unexpected toxic effect not
91 revealed by conventional approaches in the context of sediments from coastal areas, and the level of contamination
92 was not observed to be proportional to the ecotoxicological assessment. Another study on contaminated sediments
93 dredged from an urban river (New Jersey, USA) showed by calculating the Hazard Quotient that 7 years after
94 dredging, the risk associated with metals (especially Cu, Pb, and Hg) remained high, but without attention to the
95 effects of contaminants on biota and ecological functions (Soetan et al., 2022). Some studies have used chemical
96 characterisation as a first step to screen the level of contamination in sediments before implementing ecotoxicity tests
97 on the most contaminated samples (Ingersoll et al., 2000). The complementary application of chemical analyses and
98 ecotoxicological testing appears to be the best method to perform risk assessment from contaminated sediment or
99 dredged material analyses (How et al., 2023). Such a combined approach can reduce the probability of false-negative
100 results and is an opportunity for decision-making in sediment management in Europe (Heise et al., 2020).

101 In Western Europe, the Rhône River, which flows from Switzerland to France, is a notable case of an engineered
102 river that flows managed from Switzerland to France. This river corridor presents major urban-industrial
103 conurbations (mainly Lyon and Geneva) and a complex land-use history (Thorel et al., 2018). Sedimentary quality
104 issues related to Rhône River management make it a satisfactory candidate for assessing the chemical and ecological

105 risks of an on-land deposit scenario. Since 1998, numerous monitoring and restoration works have been conducted,
106 including dredging and restoration operations in the framework of the RhônEco Program (Lamouroux et al., 2015;
107 Olivier, 2016) and the Rhône sediment observatory (Piégay et al., 2022). Spatiotemporal trends of sediment
108 contamination have been extensively evaluated by dated sediment core studies on Metal Trace Elements (MTE) and
109 Persistent Organic Pollutants (POPs), such as polychlorinated biphenyls (PCBs), organochlorine pesticides, and
110 brominated flame retardants (Desmet et al., 2012; Mourier et al., 2014; Dendievel et al., 2020a; Liber et al., 2019;
111 Vauclin et al., 2021). In a synthesis work combining dated sediment cores and bed sediments, Dendievel et al.
112 (2020b) highlighted a combined toxicity risk mainly related to metals in the Upper Rhône River, particularly due to
113 copper release and leaching (vineyards and mine tailings). Downstream of the « Grand Lyon » urban area, high
114 concentrations of POPs, their metabolites, and MTE produced a major increase in the mixture risk along the Middle
115 and Lower Rhône River (Dendievel et al., 2020b).

116 This rich scientific literature has led us to develop an approach consistent with these previous studies. Thus, in this
117 study, our objective is in line with the concern of the combined analysis. We aim to characterise the potential toxicity
118 and, therefore, the risk to ecosystems, in the context of on-land deposits after dredging. Contaminant measurements
119 and bioassays were conducted on a panel of representative sediments extracted from the Rhône River in France.
120 First, the evaluation of contaminant levels (MTE, PCB) in sediments was interpreted with ecotoxic reference values
121 and calculations to estimate potential ecotoxicological risk. Acute (plant germination and earthworm avoidance) and
122 chronic bioassays (ostracod and earthworm growth and reproduction tests) were performed to evaluate the effects of
123 contaminants on ecological functions. Finally, all the results are discussed to define the potential risk and to provide
124 recommendations in the framework of a land-based sediment deposit scenario.

125 **Materials and methods**

126 *Sediment sampling*

127 Sediment surface samples were taken from four sites to represent the sediment diversity found along the
128 Rhône River. In the north, sediment was sampled from the marina of the Aix-les-Bains at Lake Bourget (LDB site;
129 45°41'38" N, 5°53'26" E) in May 2010. This site is a known hotspot for PCB contamination (Touzé and Bataillard,
130 2011; Lécrivain et al., 2018). Three other samples were collected from the Rhône River backwater areas. These areas
131 are permanently connected to the main channel of the river and continually supplied with water and sediment
132 (Desmet et al., 2012, Mourier et al., 2014). The BRE site (45°28'48"N, 4°46'83"E) was sampled in May 2008. This
133 site was carefully selected because it is located downstream of the Lyon urban area, and its industrial corridor which
134 extends south of the city. To the south, the sediment at the GEC site was collected in February 2011 (44°23'23" N;
135 4°39'21" E). This corresponds to a secondary channel located in the Pierrelatte floodplain, downstream of the
136 confluence with the Isere and Drôme Rivers. Finally, the TRS site is the most downstream site (43°43'30" N,
137 4°37'07" E), located near the Rhône River delta, and the sediment was collected in February 2012. The samples
138 were collected using a UWITEC© corer in order to obtain samples from short sediment cores. The sampling depth
139 reached was approximately 20–30 cm to fit with the most recent sediments (age-depth models are available in

140 Mourier et al., 2014). To obtain sufficient material for the tests, we repeated the operation several times over a
141 limited area at the same sampling depth. These sediments were stored in containers in a cold chamber at 5°C in the
142 dark. After storage, the decanted water was separated from the sediments. Each sediment samples in different storage
143 containers was homogenised before the experiments.

144 ***Analytical methods***

145 The physical and chemical characteristics (pH, conductivity, TOC – total organic carbon, granulometry),
146 agronomic properties (potassium, nitrogen, and C/N ratio), and pollutant contents (PCBs and trace elements) were
147 acquired for each sediment sample (see SI-1 for analysis protocol details).

148 All measurements and most of the treatments were performed in four replicates (exceptions are mentioned in the
149 tables). All reagents were of analytical grade or ultrapure water. The glassware and plastic containers used for the
150 tests were cleaned by soaking them in HNO₃ (5% v/v) for 12 hours and then rinsing them several times with
151 deionised water. Most of the physical and chemical characteristics of the sediments were obtained using several
152 protocols and standards (ISO 13878 for nitrogen determination, NFX 31-130 for cation exchange capacity [CEC],
153 ISO 10390 for pH, NFX 31-102 standards for water content [see details in SI-1]).

154 For metal quantification, 1 g of dry matter from each sediment sample was dissolved in a mixture of 2 mL HNO₃ and
155 6 mL HCl and heated in a CEM-type microwave oven (CEM/Xpress). After filtration through ash-free filter paper,
156 25 mL ultrapure water was added. Then, metal concentrations were measured by Atomic Absorption Spectrometry
157 (AAS) with a Perkin Elmer PinAAcle 900 T. Flame absorption was used for Cr, Pb, and Zn (standard FD T90-112;
158 AFNOR,1998), and a graphite furnace was employed for Cd and Cu (NF EN ISO 15586; AFNOR, 2004). The
159 detection and quantification limits (see SI-1) were calculated as described by Baffi et al. (2002). Certified Reference
160 Material (BCR-280 R, lake sediment from EU Joint Research Centre) and control samples, such as Surface Water
161 Level 2 (Spectrapure standards), were used to assess the efficiency of mineralisation and the calibration of the metal
162 studied. PCBs were analysed by the EUROFINS laboratory, Saverne, France (www.eurofins.fr) and extracted in
163 accordance with the certified method: QMA 504-192 (DIN ENISO/IEC 17025:2000; for details, see the references in
164 Sup. Materials 1, and in Mourier et al., 2014). The sample extract was analysed by high resolution gas
165 chromatography–high resolution mass spectrometry (HRGC/HRMS) with a VG-AutoSpec in selected ion
166 monitoring (SIM) mode. For the seven indicator PCBs (PCB 28, 52, 101, 118, 138, 153 et 180), the quantification
167 limits ranged from 0.042 µg/kg dry weight (DW; PCBs 118 and 180) to 0.146 µg/kg DW (PCB 153).

168 ***Sediment ecotoxicity assessment protocols***

169 Sediment ecotoxicity was assessed using acute and chronic ecotoxicity bioassays (SI-2). Acute tests
170 comprised plant bioassays (germination and early root growth) based on species that are known to be tolerant or
171 which are potential bioaccumulators of POPs and MTE. These plant species include zucchini (*Cucurbita pepo*),
172 rapeseed (*Brassica napus*), ray grass (*Lolium perenne*), and black mustard (*Brassica nigra*). These choices were
173 based on key literature showing that *Lolium perenne* (Tato et al., 2011), *Curcubitta peto* (White, 2001; Whitfield
174 Aslund et al., 2007, 2008), and *Brassica napus* (Javorska et al., 2009) potentially accumulate pollutants, especially

175 PCBs. Bedell et al. (2003; 2013) also highlighted a significant response of these plants to phytotoxicity tests conducted
176 on canal and port sediments. Moreover, some of these species or families are recommended in the international
177 standard for this test (ISO 11269-1) (SI-2). During the germination tests on microplates, the seeds were deposited on
178 blotting paper in contact with the sediment matrix. Therefore, the effect measured on germination was clearly linked
179 to the availability of water or soluble compounds via root suction. Two replicates were performed for GEC, three for
180 BER, and four for TRS and LDB, depending on the available quantity of sediment.

181 Germination and root inhibition growth tests were performed on microplates (Phytotoxkit ® supplied by R-
182 Biopharm, France). The microplates used were made of transparent plastic and were flat and shallow with two
183 compartments, one containing a solid matrix (ISO substrate or sediment) humidified at 70% retention capacity
184 (Phytotoxkit, 2004). The other compartment was empty and allowed for the emergence and growth of seedlings. The
185 cells were then incubated vertically in the dark for 48 h at 19°C in a climate culture room. The analysed sediments
186 were compared with the control (100% ISO substrate). The reference soil ISO substrate (OECD, 2010) is a mixture
187 of 10% peat, 70% silica (industrial sand smaller than 2 mm, with more than 50% of the particles comprising between
188 50 µm and 200 µm), 20% clay (kaolin content: less than 30%), and ≤ 1% CaCO₃.

189 An additional acute ecotoxicity bioassay was performed on the earthworm *Eisenia fetida* to test its avoidance (ISO
190 11268-1). It involved studying the behaviour of *E. fetida* placed in a container with two compartments: one with the
191 sediment to be tested and the other with the ISO substrate (OECD, 2010). This test was valid if the number of
192 dead/missing worms was less than 10% per treatment (SI-2).

193 The chronic tests were based on the growth and reproduction of *Eisenia fetida* (ISO 11268-2). This test was
194 performed on 10 adult earthworms placed in the sediment for 4 weeks. At the end of the experiment, adult
195 earthworms were weighed and compared with that of the control. The test was then continued for four more weeks to
196 allow reproduction, after which the juveniles were recovered, counted, and compared with the controls (see SI-2).

197 The Ostracodtoxkit® test was used (supplied by R-Biopharm, France) in order to place the ostracod *Heterocypris*
198 *incongruens* in direct contact with the sediment to be tested for 6 days (the control was performed with washed,
199 sieved, and dried sand supplied in the Ostracodtoxkit®). This test allowed to assess the growth and mortality of
200 ostracods (SI-2).

201 ***Data analysis***

202 ***Risk quotient assessment***

203 The RQ assessment is based on the measured environmental concentration (MEC) of each pollutant, divided
204 by the corresponding PNEC_{sed} (Predicted No Effect Concentration for sediment-dwelling organisms), currently used
205 in ecotoxicological studies (ECB, 2003). PNEC_{sed} values are available for a large variety of pollutants
206 (<https://substances.ineris.fr/fr/>; see Table 2). However, the PNEC_{sed} values are not defined for Cr and ΣPCBi; thus,
207 we used the threshold effect concentration (TEC) as the reference value for the RQ estimate (Table 2). Finally,
208 individual RQs (for each pollutant) were added to assess the risk of mixtures according to the Concentration

209 Addition model, which assumes a similar mode and site of toxic action of all pollutants on sediment-dwelling
210 organisms (Backhaus and Faust, 2012).

211 Mixture risk quotients (RQ_{Mix}) were calculated based on (1):

212 (1) $RQ_{\text{Mix}} = \sum (\text{MEC}_{i,x} / \text{PNEC}_{\text{sed } i})$

213 In the aforementioned equation, RQ_{Mix} corresponds to the sum of all RQs. The RQ for each pollutant is based on
214 $\text{MEC}_{i,x}$, which is the Measured Environmental Concentration of pollutant i at site \times ($\text{mg kg}^{-1} \text{ dw}$), and on $\text{PNEC}_{\text{sed } i}$
215 according to Equilibrium Partitioning ($\text{mg kg}^{-1} \text{ dw}$). RQ and RQ_{Mix} values range from < 1 (negligible risk) $>$ to > 100
216 (high risk) according to Gosset et al. (2020) and Perrodin et al. (2012).

217 **Statistical tests**

218 The significance level of the differences between the bioassay data for different sediments and groups of
219 stations was assessed using the t test and Mann–Whitney U-test. Statistical significance was set than 0.05. Statistical
220 analyses were performed using STATISTICA© (version 10).

221 Post hoc comparisons between normally distributed populations (*e.g.* germination and root inhibition (RI) at 7 days
222 on microplate assays) were performed using the t test. A non-parametric test (Mann-Whitney U-test) was applied
223 when a normal distribution could not be achieved. The Mann-Whitney test was used to analyse (i) the differences in
224 the distribution of earthworms in the two compartments at the end of the avoidance test and (ii) between the control
225 and the sediment in the growth and death of ostracods and earthworms (reproduction test).

226 **Results/Discussion**

227 **Sediment characterisation**

228 The sediments tested were slightly alkaline (pH 7.5; Table 1). The conductivity measurements showed two
229 groups: (i) TRS and BER sediments between 307 and 347 $\mu\text{s/cm}$, and (ii) LDB and GEC sediments between 644 and
230 795 $\mu\text{s/cm}$ (Table 1). These groups can be linked to local physical and geochemical settings or to potential release in
231 nearby areas. Overall, the sediment grain size was silty to silty-sandy, with the highest percentage of clays for TRS
232 and GEC sediments (10 to 8%). The Total Organic Carbon content (TOC) was between 12.8 and 17.9 mg/kg DW for
233 most of the sediment samples, except for that of LDB, which was three times higher than the others (68.2 mg/kg
234 DW). LBD sediment also had much higher potassium and nitrogen contents than the other sediments (Table 1).
235 Phosphorous contents ranged from 0.13 to 0.24 g/kg DW . The C/N ratio was approximately 12, except for that of
236 BER sediment, which is approximately 9.7 (Table 1). These values are not observed to be limiting factors for plant
237 growth and development (Doucet, 1992). Only BER and GEC sediments had nutrient contents (especially,
238 potassium) that were slightly lower than those of the other sediments (Table 1).

239 **Trace Metal Elements (TME) and $\sum\text{PCBi}$ contents**

240 Concerning TME, Cd presents the most severe concentrations for all sediments: between 0.9 and 1.4 mg/kg
241 DW (Table 1). Such concentrations often reach the Probable Effect Concentration threshold ($\text{PEC} = 0.99 \text{ mg/kg}$

242 DW) defined for sediment-dwelling organisms (MacDonald et al., 2000a) but not that of the French regulatory
243 threshold S1, defining the contaminant levels to be considered in sediments extracted from rivers and canal dredged
244 sediments (JORF, 2006; see also Table 2). Because of the variability of the measures, the contents of Cr and Ni were
245 also high for the different samples (47.7–70.2 mg/kg DW for Cr, and 23.5–34.1 mg/kg DW for Ni). Even if these
246 values are higher than the local geochemical background along the Rhône River, they are much lower than the
247 French threshold S1 (Table 2). For the other elements, the concentrations of Cu, Pb, and Zn were generally higher at
248 LDB and GEC site than at the other sites (two to three times higher). Cu and Zn were lower than the French
249 threshold S1 (JORF, 2006). Pb values in the LDB sediment slightly exceeded the regulatory threshold S1 (Tables 1
250 and 2).

251 The sediment concentrations in the sum of the seven PCB indicators (Σ PCBi) were highest for LDB site (1204 μ g/kg
252 DW on average). This value largely exceeds the regulatory threshold S1 (680 μ g/kg) and PEC for total PCBs
253 (MacDonald et al., 2000a). GEC sediment also contained 122.1 μ g/kg DW Σ PCBi on average. This concentration
254 was between the TEC (35 μ g/kg; MacDonald et al., 2000b) and the PEC/S1. Finally, BER and GEC sediment
255 contained between 22 and 27 μ g/kg DW of Σ PCBi on average. Even if these levels are lower than the
256 aforementioned thresholds (TEC, PEC, S1), they remain within the ranges of the sediment benchmarks (10 μ g/kg
257 and 60 μ g/kg) calculated from biota-to-sediment accumulation factor models in fish (Babut et al., 2012; Lopes et al.,
258 2012).

259 In the studied samples, PCB congener concentrations were in the order of PCB 153 > PCB 180 > PCB 138 > PCB
260 101 (Table 1). This assemblage might be considered a marker related to river contamination by industries using
261 PCBs, as demonstrated by Lasserre et al. (2009). Mourier et al. (2014) highlighted the same congener distribution in
262 historical sediment cores extracted along the Rhône River (1960–2011).

263 In summary, the sediment samples present decreasing concentrations of TME and PCB in the following order: LDB
264 > GEC > TRS > BER. Except for the high levels of contaminants measured in the LDB sediment, the "hazard status"
265 of the other sediments is complicated to define because some contaminants have high concentrations individually.
266 They all present a potential risk of pollutant transfer to plants, and an ecotoxicological risk for organisms likely to
267 colonise and reproduce in such deposits. Based on their physicochemical characteristics and agronomic values, the
268 sediment samples are not expected to be a limiting factor for plant growth.

269 ***Potential environmental risk***

270 To evaluate the ecotoxicological risk and potential hazard of the sediment samples, we calculated the risk
271 quotient mixture (RQ_{mix}) based on MEC and $PNEC_{\text{sed}}$ data (see section 2.4.1). In our study, the mixture quotient was
272 based on the concentrations of metals and PCBs to assess the relative contribution of each pollutant to the cocktail
273 effect on sediment-dwelling organisms.

274 LDB sediment presents the most important concentrations of trace elements and Σ PCBi (Table 1). Consequently,
275 these concentrations lead to a high RQ_{Mix} value (> 100), especially driven by Cu and Σ PCBis. Individual RQ for Cr,
276 Ni, Pb, and Zn suggested a low-level risk for the mixture. RQ values derived from Cd appear relatively low owing to

277 the high $PNEC_{sed}$ limit (2.5 mg/kg), which underestimates the risk. This result is somewhat surprising because Cd
278 values are problematic with respect to toxicology thresholds for most sites (Table 2: values > PEC). The other tested
279 sediments (from BRE, GEC, and TRS) presented a moderate ecotoxicological risk ($38.17 < RQ_{Mix} < 56.51$). The
280 sediments tested can be classified according to a decreasing gradient of the RQ_{Mix} total in the following order: LDB
281 > GEC > TRS > BER.

282 The Risk Quotient mixture (RQ_{Mix}) is based on the analysed metals and PCBs to assess the relative contribution of
283 each pollutant to the effect of a cocktail of contaminants on sediment-dwelling organisms. According to data in the
284 literature on core sediments from the Rhône River, the cumulative ecotoxicity risk is mainly driven by (1)
285 dichlorodiphenyltrichloroethane (DDT) metabolites (DDE and DDD), (2) lindane isomers (HCHs), and (3) PCBs
286 that remain a major cause of concern for the Rhône (Dendievel et al., 2020a). The Risk Quotient mixture could also
287 fluctuate, such as in English coastal rivers (Manuel Nicolaus et al., 2015), or negligible, such as in the Garonne
288 Estuary (metals: Larrose et al., 2010), due to sorption processes and organic matter degradation, reducing the
289 estimation of the impact of industrial inputs in sediments, while high metal concentrations have been found in fauna
290 (Lanceleur et al., 2011). Other studies have been shown that in such a global approach by risk ratio, or RQ, this is
291 mainly due to the acetochlor ($C_{14}H_{20}ClNO_2$) in four sediments of Argentina's rural streams (Fernández San Juan et
292 al., 2022) or to the tributyltin concentrations at almost all the study sites of the Odra River estuary (Kucharski et al.,
293 2022). Such results cited also underly the weight of POP contamination in such an evaluation. At the scale of the
294 French Rhône River (Figure 2), the data presented in this work are consistent with other RQ_{Mix} values published for
295 sediment cores (Dendievel et al., 2020b). Thus, the BER, GEC, and TRS RQ_{Mix} values are in line with the known
296 RQ_{Mix} values in the Middle and Lower Rhône River sections (Figure 2). LDB sediment is an exception and offers
297 strong RQ_{Mix} without link between ETL and MTE. Notably, LDB sediments are from the Aix-les-Bains marina,
298 located in Bourget Lake. The latter lake is a major sediment deposition zone, only temporarily connected to the
299 Rhône River by a short channel ("canal de Savières") during floods, which explains no relationship between the
300 LDB peak and the other upstream sites.

301 **Ecotoxicity evaluation**

302 Our ecotoxicological approach used acute (germination, avoidance) and chronic exposure tests (e.g.
303 reproduction, growth) on different components of an ecosystem, such as primary producers (plants), decomposers
304 (earthworms), and target organisms of the deposit (plants) or the sediment (ostracods).

305 *Acute toxicity tests of sediments*

306 **Early germination and growth on sediment in microplates**

307 Germination tests using the French HP14 methodology were performed on four sediments. The germination
308 inhibition of *Cucurbita pepo* ssp. *pepo* was more notable for GEC and LDB (ca. 18%) than for BER and TRS (3.3 to
309 5%; Table 3). This result is much less than that in a study analysing the germination of *Cucurbita pepo* ssp. *pepo* in
310 soil amended with DDT (~1500 ng/g), which showed the development of 60% of the seeds (n=5; Whitfield Ashlund

311 et al., 2010). Three sediments (from BER, TRS, LDB) were also tested with *Brassica nigra* and *Lolium perenne*, and
312 germination inhibition seemed effective for LDB (both plants: 1 to 4%) and BER (*B. nigra*, ca. 10%) sediments. By
313 contrast, an inhibition of the germination associated with a notable standard error on TRS suggested a lack of effect
314 or a weak stimulation of the seeds. Chigbo and Batty (2013) reported that the combined effect of Cr and
315 Benzo(a)pyrene (BaP) significantly reduced the germination rate of *L. perenne* in comparison of the control and
316 assays with single contaminant effects. These variations in *L. perenne* germination (e.g. inhibition or stimulation)
317 have also been detected using microplate assays and sediments (Bedell et al., 2003; 2013). For *Brassica napus*, the
318 two sediments tested (from TRS and LDB) had no significant effect on seed germination (Table 3). This result
319 supports that of Wierzbicka and Obidzinska (1998), who showed that seed coat morphology is an important defence
320 against some metallic ions, especially for Brassicaceae (Cruciferae). However, even if *B. napus* has high emergence
321 variability (Fernandez et al., 2005), the sensitivity of germination of *B. napus* to dredged canal sediment was
322 observed during a three-year phytoremediation trial (King et al., 2006). Regarding other plants that could be used in
323 these tests, alfalfa (*Medicago sativa*), a dicotyledonous species, seems to be the most relevant species for screening
324 sediment toxicity before reuse (Lecomte et al., 2019). Such variability and sensitivity according to the species tested,
325 which was also observed in our data, may result from the different protection provided to embryos by seed coverage
326 or the permeability of this coverage to contaminants or water.

327 Two physiological phenomena are active during germination: oxygen use increases when respiration begins and
328 imbibition. Imbibition can occur when the seed rehydrates but can also be linked to sediment properties such as
329 texture (and clay content). This case was not observed in our study because BER sediment has greater inhibition of
330 root growth and germination than TRS sediment, despite a lower clay content for the former than for the latter (4%
331 vs 10%; Table 1). Another hypothesis suggests that differences in seed germination are related to seed respiration
332 and that levels of metals (e.g. As, Hg, or Cd) and PCBs are potential inhibitors of seed respiration (Sethy and Ghosh,
333 2013). Beyond contaminants, these variations can be attributed to several factors related to the experimental design,
334 such as the use of microplates (blotting paper) and seed quality.

335 Germination tests performed on seeds with decant water collected from the sediments did not show a significant
336 inhibitory effect on germination or early root growth. However, a correlation was observed between the root growth
337 of black mustard and the concentrations of pollutants in the decanted water of sediment BER (Fisher test = 0.0103 at
338 $P < 0.05$). Thus, according to our results, the sediment matrix is more relevant for toxicity tests than the decant
339 water. Such conclusions have also been well demonstrated in other studies, highlighting that sediment induces a
340 higher toxic response than pore water does (Palma et al., 2014). In our case, we found correlations between
341 *Cucurbita pepo* germination and RI with sediment conductivity (r^2 of 0.77 and 0.92, respectively), suggesting an
342 influence of water quality on seed imbibition and, therefore, germination.

343 Notably, in the case of germination inhibition, root growth inhibition generally occurs for a given type of sediment
344 and plant species and *vice versa* (Table 3). Regarding the inhibition of early root growth (see SI-2 for details), the
345 radicles of *Cucurbita pepo* were inhibited in all tested sediments (from TRS, GEC, BER, and LDB), and

346 considerable inhibition was observed for *Lolium perenne* (from TRS, BER, and LDB; Table 3). The radicle growth
347 of *Brassica napus* and *B. nigra* was neither affected nor slightly stimulated (Table 3). However, for *Lolium perenne*,
348 opposing effects were observed in our results: germination inhibition and increased root growth (Table 4). This
349 contradiction was also highlighted by Chigbo and Batty (2013), who demonstrated that root and shoot elongation of
350 *L. perenne* was significantly inhibited ($p > 0.05$) at a high concentration of Cr, whereas increasing concentrations of
351 BaP accelerated shoot elongation. In our case, RI was more pronounced for *Lolium perenne* in BER sediment than in
352 TRS sediment, despite similar Σ PCBi contents. BER sediment had lower metal contents than TRS sediment did
353 (Table 1). Consequently, we concluded that there was no correlation between germination inhibition and early root
354 growth. A similar absence of proven correlations between pollutant content and the effect observed in this type of
355 acute test was observed by Mamindy-Pajany et al. (2011) in some bioassays with Poaceae (*Sorghum saccharatum*)
356 and Brassicaceae (*Lepidium sativum*) on seaport sediments. Nonetheless, in the case of the most contaminated
357 samples (e.g. LBD and GEC sediments), *Cucurbita pepo* and *Lolium perenne* were the species associated with the
358 most marked inhibition.

359 Finally, acute germination and early root growth tests allowed us to classify the sediments from most to least
360 inhibition regarding the plants tested: LDB > BER \geq GEC > TRS. This order is not only related to the levels of
361 TMEs and PCBs in the sediment but also to the availability of nutrients, such as nitrogen and phosphorus, which
362 depend on the sediment type. The presence of other POPs, which were not measured in this study, is also a potential
363 factor affecting plants. Because of the experimental system used (blotting paper), The TMEs were undoubtedly the
364 only elements that had an effect in both experiments.

365 **Earthworm (*Eisenia fetida*) avoidance test**

366 The earthworm avoidance test requires a large quantity of sediment; thus, it was only conducted on BER,
367 LDB, and TRS sediments (Table 4). We observed that from 60 to 62.5% of the earthworms preferred LDB sediment
368 to the ISO reference soil. By contrast, earthworms avoided TRS and BER sediments (Table 4). These differences can
369 be explained by the organic matter and clay content (Table 1). LDB sediment had a higher organic matter content
370 than TRS and BER sediments did (Table 1), which may explain the earthworms' preference for LDB sediment,
371 despite the samples being contaminated by Pb and PCBs (Table 1). Davies et al. (2003 a,b) demonstrated the effect
372 of Pb on worms, especially through accumulation due to uptake regulation at low contamination levels. Moreover,
373 earthworms can sense chemicals via many chemoreceptors, as shown by the contamination avoidance results
374 obtained for pesticides (Zhou et al., 2007; Garcia et al., 2008) and 2,4,6-trinitrotoluene (TNT) (Schaefer, 2004).
375 Edwards and Bohlen (1996) observed that most organophosphates (OP) herbicides were not toxic to *E. fetida* worms
376 because they were unable to transform OP into toxic metabolites. Although toxic, substances such as cadmium salts
377 may not be perceived as repulsive if the body is forced to remain in contact with them (Greenslade and Vaughan,
378 2003). An avoidance test performed on four fluvial sediments showed 47.2% of earthworms' avoidance and 45%
379 for dam sediment avoidance (Lecomte et al., 2019).

380 TRS sediment was richer in clay (10%) than BER and LDB sediment. This clay content may be one of the factors
381 explaining sediment colonisation by earthworms. Hund-Rinke et al. (2005) found different avoidance factors (EC50)
382 according to the soil type in the presence of pentachlorophenol (PCP): 8 mg/kg for sandy soil and 24 mg/kg for silty
383 soil. This result clearly illustrates the key role of the sediment characteristics with respect to the real conditions of
384 exposure of earthworms to contaminants, such as through the adsorption of PCP, which is positively correlated with
385 organic matter. Finally, the sediments were graded from least to most colonisable by earthworms in the following
386 order: BER – TRS – LDB. The avoidance test remains a satisfactory compromise for the rapid, easy evaluation of
387 potential colonisation by earthworms.

388 *Chronic toxicity tests on sediments*

389 **Monitoring mortality and growth of *Heterocypris incongruens*: ostracod test**

390 In this section, we aim to determine the growth inhibition and mortality of the benthic ostracod
391 *Heterocypris incongruens*. To achieve this objective, we use a chronic sediment toxicity test (Ostracodtoxkit®) in
392 the presence of the sediment samples and compare them to a control (sand provided in the kit). The maximum
393 ostracod mortality rate was approximately 8% in the six control replicates. This low percentage (<20%) and the
394 correct growth of the ostracod length in the controls (x1.5 between T0 and T+6) validated the experiment.

395 Regarding the tests performed on the Rhône sediments, almost all ostracods in contact with the LDB sediment died
396 (90%), and a mortality of 50% was reported for ostracods in contact with the GEC sediment (Figure 1-A). Thus,
397 LDB and GEC sediment had a major effect on *Heterocypris incongruens* survival, even if the variability associated
398 with this test was greater for LDB and GEC sediment than for the other sediments. Ostracod growth was inhibited
399 from 35 to 110% in the four sediments tested (Figure 1-B). Their growth was considerably inhibited by LDB
400 sediment and to a lesser extent by GEC sediment. However, TRS and BER sediment showed low effect on ostracod
401 survival but a significant effect (40 to 50% on average) on the growth of *H. incongruens*. A strong relationship can
402 be assumed between the pollutant concentrations in the sediments (Table 1), and the effects observed on ostracods.
403 High correlations ($r^2 > 0.8$) were observed among physicochemical parameters (e.g. PCBi levels and several TMEs,
404 including Cu, Pb, and Zn) and growth inhibition or mortality.

405 Our results agree with those of another study, which examined 33 sediments and concluded that the ostracod
406 microbiotest is a reliable, sensitive alternative to whole sediment assays for toxicity assessment (Belgis et al., 2003).
407 Pesce et al. (2020) recently highlighted the promising prospects of the ostracod test for investigating the
408 ecotoxicological effects of metal contamination on natural sediment communities, using the degradation and
409 decomposition of particulate organic matter as a functional descriptor. Lecomte et al. (2019) demonstrated up to a
410 40.6% increase in ostracod growth inhibition in dam sediments but found no effects on mortality. In contrast with
411 results of Lecomte et al. (2019), our study of river sediments showed that the responses of *H. incongruens* revealed a
412 potential toxicity gradient (with mortality) in the sediments tested. In a similar context (reservoir sediments), Palma

413 et al. (2014) showed that *H. incongruens* is sensitive to contaminants found in sediments with higher mortality but
414 lower growth inhibition for similar sediment characteristics and similar MTE contents.

415 Finally, the sediments were classified according to the inhibition of the growth of *Heterocypris incongruens*: LDB >
416 GEC > TRS > BER. In the case of an on-land deposit scenario, LDB sediment, and to a lesser extent GEC sediment,
417 presented the greatest risk of toxicity for *Heterocypris incongruens*.

418 **Chronic earthworm (*Eisenia fetida*) reproduction test in TRS and LDB sediments**

419 Chronic reproductive tests of *Eisenia fetida* were performed after 4 and 8 weeks of exposure to TRS and
420 LDB sediments. After 4 weeks, the growth of 10 adult earthworms on the TRS sediment presented a loss of biomass
421 of approximately 20% (Table 5). This loss was most significant for the LDB sediment (36% of biomass loss).
422 However, this inhibition did not increase the mortality rate (Table 5). Lecomte et al. (2019) also showed a small
423 effect on biomass intake but a greater effect on earthworm reproduction with undiluted sediment. Spurgeon and
424 Hopkin (1996) demonstrated a major link among metals, especially Zn, and the survival and growth of *Eisenia fetida*
425 in artificial soil. They found a significant worm mortality gradient in soils containing from 1200 to 2000 mg.kg⁻¹ Zn
426 (all worms died in the latter case). These results were obtained for soil that was approximately 10 times more
427 polluted than our sediments. In another study (Schaefer, 2004), biomass loss reached from 10 to 20% in soils
428 contaminated with TNT. However, the author did not draw a conclusion regarding the significance of toxicity
429 because all the individuals in all the variants had lost weight by the end of the test (Schaefer, 2004). However, in
430 other soils (Luo et al., 2014), the weight loss of earthworms was significantly and positively correlated with water-
431 extractable Pb but negatively correlated with CEC. Luo et al. (2014) also observed that a low organic matter content
432 in soils could have an adverse effect on earthworm growth. In our study, CEC, Pb concentration, and TOC in the
433 LDB sediment were higher than those in the TRS sediment, which might explain the higher earthworm biomass loss
434 in LDB than in TRS (Table 1).

435 After 8 weeks, the growth inhibition affected 50% of the juveniles from TRS and LDB sediment compared to the
436 control (Table 5). In other contexts (chlorpyrifos-contaminated soils), Zhou et al. (2007) also showed a significant
437 effect of exposure to chlorpyrifos-contaminated soils on earthworm reproduction. The number of juveniles was
438 significantly and positively correlated with pH, Ca, silt content, Zn, Cd, and CEC but significantly and negatively
439 correlated with sand content (Luo et al., 2014). In our sediment samples, those from LDB and TRS presented small
440 differences in pH, silt, and sand values (Table 1). However, the high CEC, Zn, and Cd contents in LDB sediment can
441 explain the greater reduction in the number of juveniles than in TRS sediment (Tables 1 and 5). Moreover,
442 discoloration of the juveniles was observed during the experiments with LDB sediment but not with TRS sediment
443 (data not shown). The CEC and pollutant content can explain the effect of our sediments on earthworm reproduction.
444 Similarly, Hankard et al. (2005) showed that earthworms could survive in all the urban soils they tested, but
445 differences occurred in the reproduction rate, possibly affecting the fitness of worms inhabiting the sediment deposit.

446 Risk evaluation discussion

447

448 In this study, hazard characterisation (via chemical analyses) and ecotoxicological tests were conducted on a
449 panel of sediments with different levels of pollutants. Based on RQ_{mix} and a comparison of concentrations with
450 ecotoxicological thresholds, we highlighted a decreasing gradient of potential danger in the following order: LDB >
451 GEC > TRS > BER. These chemical analyses showed that LDB sediment was problematic for management in an on-
452 land deposit scenario, and the hazard was considered moderate for the other samples. The mortality and growth of
453 *Heterocypris incongruens* were well related to pollutant loads (PCBi and TMEs, especially copper, lead, and zinc),
454 but other ecotoxicological tests have provided less marked results.

455 - The acute ecotoxicity bioassays showed the effect of contaminated sediments on germination and early root
456 growth, with the greatest sensitivity for the species *Lolium perenne* and *Cucurbita pepo*. LDB, GEC, and
457 BER sediment were the most ecotoxic owing to their pollutant loads. Moreover, LDB sediment was twice
458 as rich in organic matter as the samples from other sites.

459 - The chronic tests showed significant toxicity for *Eisenia fetida* and *Heterocypris incongruens* living in LDB
460 sediment. Tests on earthworms (*E. fetida*) also demonstrated a significant effect of TRS and LDB sediment
461 on reproduction and growth. LDB sediment appeared to be toxic to ostracods (*H. incongruens*), but the
462 variability observed did not permit a definitive conclusion on its toxicity.

463 In their study of seaport sediments, Mamindy-Pajany et al. (2011) showed that the relationship between chemical
464 data and toxicity tests was unclear. Our results indicate satisfactory complementarity of the chemistry and ostracod
465 test results for all tested sediments and observed results for the other bioassays.

466 The combination of ecotoxic results and RQ_{mix} values at different concentrations in the sediments demonstrates that
467 LDB (marina) sediment is still not available for an on-land deposit scenario. TRS and GEC sediment are not directly
468 reusable, especially based on the results of chronic tests. However, BER sediment did not show a combined effect
469 and is suitable for such a scenario.

470 Moreover, the choice of scenario for the management of the sediment can be linked to other possibilities of the
471 sediment valorisation. Some studies have suggested the valorisation of sediments as a plant substrate (Ferrans et al.,
472 2022), metal recovery in a cost-integrated approach, or a circular management approach of dredged sediments
473 (Crocetti et al., 2022; Svensson et al., 2022). The latter case points to the critical and current need to consider metals
474 and POPs together, and to integrate physical and socio-environmental parameters in order to assess over time the
475 toxicity of sediments along large and heterogeneous rivers (Beyer et al., 2014; Kortenkamp et al., 2019).

476 Notably, the risk of dredged sediment is also linked to deposit site specificities. Thus, a collaborative effort among
477 stakeholders, scientists, and representatives of the government must be realised for site determination linked to risk
478 evaluation. For example, a collaborative approach among the French Government, the “Grand Port Maritime de
479 Rouen”, and a scientific committee has made it possible to define an integrative protocol to survey all the
480 compartments that might be affected by dumping in the management of port dredged sediment in the Bay of Seine
481 (France) (Marmin et al., 2014). All these monitoring procedures (i.e. sediment and water quality, including chemical

482 contamination; bioaccumulation and ecotoxicology on target species; microbiology, invertebrates, and fish surveys;
483 impact on the Natura 2000 areas) were conducted by partners under the responsibility of the “Grand Port Maritime
484 de Rouen” from 2012 to 2013 (Marmin et al., 2014).

485 **Conclusion**

486 The ecotoxicological risks induced by a scenario of land-based deposition of dredged sediments from the
487 Rhône River were assessed by characterising sediments and the associated pollution (physical and chemical
488 contents) and combining multiple tests on various organisms likely to colonise the deposit (e.g. plants, ostracods,
489 worms). The potential danger of the selected sediments was measured by simulating the risks from exposure to the
490 selected of pollutants based on the RQ_{mix} calculations. According to this approach, one site presented problematic
491 pollution for management in an on-land deposit scenario: the LDB sediment extracted from Bourget Lake Marina.
492 Various complex responses were found for the other tested sediments and sites, which confirms the need to perform
493 complementary ecotoxicological tests.

494 Bioassays involving earthworms are essential because they show that sediments can have different effects depending
495 on the exposure time. The LDB sediment had the greatest inhibitory effect on earthworm reproduction and growth.
496 In addition, some tests are sometimes difficult to interpret because earthworms seem to prefer sediments with
497 agronomic characteristics favourable to their development (high organic matter and low clay content), even though
498 this matrix may have notable toxic effects. However, our series of bioassays showed that all the sediments tested
499 were potentially ecotoxic to organisms that sought to colonise the deposit. Because it is the most toxic, LDB
500 sediment is complicated to manage; thus, the on-land deposit scenario should be avoided in this case.

501 The response to toxicity bioassays obtained on the other sediments also showed the limitations of these tests and the
502 need for a complementary assessment, particularly over time, of the dispersion, or mobility of contaminants for
503 definitive decision-making. Then, TRS sediment is not available directly for such deposits based on chronic test
504 results, and on the other hand, BER and GEC sediment can be used for such deposit scenarios with attention to
505 germination in the case of the revegetating approach. Further investigations before management should be performed
506 by a temporal simulation of ageing (by leaching in columns) or by mimicking several redox conditions.

507 Finally, an integrative approach among the physicochemical interrelations and ecotoxicological tests provided
508 relevant perspectives for management and modalities for land deposit scenarios of dredged sediments. Therefore, to
509 manage sediments dredged from the Rhône River and to deposit them on land, determining the cumulative risks
510 linked to the sediments is necessary. In addition, the content and release of pollutants from runoff water should also
511 be studied because such runoff water could be a vector of the transport and diffusion of the pollution and ecotoxic,
512 particularly for local aquatic and wetland ecosystems.

513 **Acknowledgements**

514 This project was funded by the “Rhône–Mediterranean and Corsica Water Agency”, the Auvergne-Rhône–Alpes and
515 Provence–Alpes–Côte d'Azur Regions, as well as the “Compagnie Nationale du Rhône” (CNR) in the context of the
516 “Rhône Sediments Observatory” (OSR). We thank Myriam Hammada and Marc Danjean (LEHNA Laboratory,
517 ENTPE, IAPHY team) for their technical help with sampling and during the analysis.

518

519 **References**

520 Adam G, Duncan H (2002) Influence of diesel fuel on seed germination. *Environ Pollut* 120 : 363-370.
521 [https://doi.org/10.1016/S0269-7491\(02\)00119-7](https://doi.org/10.1016/S0269-7491(02)00119-7)

522 AFNOR, 1998. Qualité de l'eau – Dosage de huit éléments métalliques (Mn, Fe, Co, Ni, Cu, Zn, Ag, Pb)
523 par spectrométrie d'absorption atomique dans la flamme (FD T90-112).

524 AFNOR, 2004. Qualité de l'eau – Dosage des éléments traces par spectrométrie d'absorption atomique en
525 four graphite (NF EN ISO 15586).

526 Alvarenga P, Palma P, de Varennes A, Cunha-Queda A (2012) A contribution towards the risk assessment
527 of soils from the São Domingos mine (Portugal): chemical, microbial and ecotoxicological indicators.
528 *Environ Pollut* 161: 50-56. <https://doi.org/10.1016/j.envpol.2011.09.044>

529 Babut M, Lopes C, Pradelle S, Persat H, Badot P-M (2012) BSAFs for freshwater fish and derivation of a
530 sediment quality guideline for PCBs in the Rhône basin, France. *J Soils Sediments* 12(2): 241–251.
531 <https://doi.org/10.1007/s11368-011-0448-y>

532 Backhaus T, Faust M (2012) Predictive environmental risk assessment of chemical mixtures: a conceptual
533 framework. *Environ Sci Technol* 46 : 2564–2573. <https://doi.org/10.1021/es2034125>

534 Baffi C, Bettinelli M, Beone GM, Spezia S (2002) Comparison of different analytical procedures in the
535 determination of trace elements in lichens. *Chemosphere* 48: 299-306. [https://doi.org/10.1016/s0045-](https://doi.org/10.1016/s0045-6535(02)00094-2)
536 [6535\(02\)00094-2](https://doi.org/10.1016/s0045-6535(02)00094-2)

537 Bedell J-P, Bazin C, Sarrazin B, Perrodin Y (2013) Assessment of the phytotoxicity of seaport sediments
538 in the framework of a quarry deposit scenario: germination tests on sediments aged artificially by column
539 leaching. *Arch Environ Contam Toxicol.* 65 (1): 1-13. <https://doi.org/10.1007/s00244-013-9881-9>

540 Bedell J-P, Briant A, Delolme C, Perrodin Y (2003) Evaluation of the phytotoxicity of contaminated
541 sediments deposited on soil. I. Impact of water draining from the deposit on the germination of
542 neighbouring plants. *Chemosphere* 50: 393-402. [https://doi.org/10.1016/s0045-6535\(02\)00600-8](https://doi.org/10.1016/s0045-6535(02)00600-8)

543 Belgis ZC, Persoone G, Blaise C (2003) Cyst-based toxicity tests XVI--sensitivity comparison of the solid
544 phase *Heterocypris incongruens* microbiotest with the *Hyalella azteca* and *Chironomus riparius* contact
545 assays on freshwater sediments from Peninsula Harbour (Ontario, Canada). *Chemosphere* 52(1): 95-101.
546 [https://doi.org/10.1016/S0045-6535\(03\)00186-3](https://doi.org/10.1016/S0045-6535(03)00186-3)

547 Beyer J, Petersen K, Song Y, Ruus A, Grung M, Bakke T, Tollefsen KE (2014) Environmental risk
548 assessment of combined effects in aquatic ecotoxicology: A discussion paper. *Mar Environ Res* 96, 81–91.
549 <https://doi.org/10.1016/j.marenvres.2013.10.008>

550 Buch AC, Brown GG, Niva CC, Sautter KD, Sousa JP (2013) Toxicity of three pesticides commonly used
551 in Brazil to *Pontoscolex corethrurus* (Müller, 1857) and *Eisenia andrei* (Bouché, 1972) *Appl Soil Ecol*
552 69:32-38. <https://doi.org/10.1016/j.apsoil.2012.12.011>

553 Buitrago BH, Muñoz PF, Ribé V, Larsson M, Engwall M, Wojciechowska E, Wazara S (2013) Hazard
554 assessment of sediments from a wetland system for treatment of landfill leachate using bioassays.
555 *Ecotoxicol Environment Saf* 97: 255-262. <https://doi.org/10.1016/j.ecoenv.2013.08.010>

556 Chen YX, Zhu GW, Tian GM, Zhou GD, Luo YM, Wu SC (2002) Phytotoxicity of dredged sediment
557 from urban canal as land application. *Environ Pollut* 117: 233–241. [https://doi.org/10.1016/S0269-7491\(01\)00270-6](https://doi.org/10.1016/S0269-7491(01)00270-6)

559 Chigbo C, Batty L (2013) Effect of combined pollution of chromium and benzo (a) pyrene on seed growth
560 of *Lolium perenne*. *Chemosphere* 90: 164-169. <https://doi.org/10.1016/j.chemosphere.2012.06.021>

561 Coehlo C, Foret C., Bazin C, Leduc L, Hammada M, Inacio M, Bedell J-P (2018) Bioavailability and
562 bioaccumulation of heavy metals of several soils and sediments (from industrialized urban areas) for
563 *Eisenia fetida*. *Sci Tot Environ* 635: 1317-1330. <https://doi.org/10.1016/j.scitotenv.2018.04.213>

564 Crocetti P, Gonzalez-Camejo J, li K, Foglia A; Eusebi AL, Fatone F (2022) An overview of operations
565 and processes for circular management of dredged sediments. *Waste management* 146: 20-35.
566 <https://doi.org/10.1016/j.wasman.2022.04.040>

567 Czerniawska-Kusza I, Ciesielczuk T, Kusza G, Cichon A (2006) Comparison of phytotoxkit microbiotest
568 and chemical variables for toxicity evaluation of sediments. *Environ Toxicol* 21(4): 367-372.
569 <https://doi.org/10.1002/tox.20189>

570 Czerniawska-Kusza I, Kusza G (2011) The potential of the phytotoxkit microbiotest for hazard evaluation
571 of sediments in eutrophic freshwater ecosystems. *Environ Monit Assess* 179: 113-121.
572 <https://doi.org/10.1007/s10661-010-1722-y>

573 Da Luz TN, Ribeiro R, Sousa JP (2004) Avoidance tests with Collembola and earthworms as early
574 screening tools for site-specific assessment of polluted soils. *Environ Tox Chem* 23, 2188-2193.
575 <https://doi.org/10.1897/03-445>

576 Davies NA, Hodson ME, Black S (2003a) Is the OECD acute worm toxicity test environmentally
577 relevant? The effect of mineral form on calculated lead toxicity. *Environ Pollut* 121(1): 49-54.
578 [https://doi.org/10.1016/s0269-7491\(02\)00206-3](https://doi.org/10.1016/s0269-7491(02)00206-3).

579 Davies NA, Hodson ME, Black S (2003b) The influence of time on lead toxicity and bioaccumulation
580 determined by the OECD earthworm toxicity test. *Environ Poll* 121: 55-61. [https://doi.org/10.1016/s0269-7491\(02\)00207-5](https://doi.org/10.1016/s0269-7491(02)00207-5)

582 Dendievel A-M, Mourier B, Coynel A, Evrard O, Labadie P, Ayrault S, Debret M, Koltalo F, Copard Y,
583 Faivre Q, Gardes T, Vauclin S, Budzinski H, Grosbois C, Winiarski T, Desmet M (2020a) Spatio-
584 temporal assessment of the polychlorinated biphenyl (PCB) sediment contamination in four major French
585 river corridors (1945–2018). *Earth Syst Sci Data*, 12, 1153–1170. <https://doi.org/10.5194/essd-12-1153-2020>

587 Dendievel A-M, Mourier B, Dabrin A, Delile H, Coynel A, Gosset A, Liber Y, Berger J-F, Bedell J-P
588 (2020b) Metal pollution trajectories and mixture risk assessed by combining dated cores and subsurface
589 sediments along a major European river (Rhône River, France). *Environ Int* 144: 106032.
590 <https://doi.org/10.1016/j.envint.2020.106032>

591 Desmet M, Mourier B, Mahler B J, Van Metre PC, Roux G, Persat H, Lefèvre I, Peretti A, Chapron E,
592 Simonneau A, Miège C, Babut M (2012) Spatial and temporal trends in PCBs in sediment along lower
593 Rhône River, France. *Sci Tot Environ* 433, 189-197. <https://doi.org/10.1016/j.envint.2020.106032>

594 Doucet, R 1992 *La Science Agricole – Climat, sols et productions végétales du Québec*. Cégep Joliette –
595 De Lanaudière. Éditions Berger, Québec. ISBN : 9782921416085

596 ECB, 2003. Technical Guidance Document (TGD) for risk assessment in support of Commission
597 Directive 93/67/EEC on Risk Assessment for new notified substances Commission Regulation (EC) No
598 1488/94 on Risk Assessment for existing substances Directive 98/8/EC of the European Parliament and of
599 the Council concerning the placing of biocidal products on the market. European communities, Ispra
600 (Italy).

601 Edwards C.A., Bohlen P.J. (1996). Biology of earthworms. 3rd ed. Chapman and Hall, London.

602 Eijsackers H (2010) Earthworms as colonizers: primary colonization of contaminated land, and sediment
603 and soil waste deposits. *Sci Total Environ* 408: 1759-1769.
604 <https://doi.org/10.1016/j.scitotenv.2009.12.046>

605 Eppe G, Focant JF, Pirard C, Xhrouet C, Maguin-Rogister G, De Pauw E (2006) Analyse des dioxines par
606 Chromatographie Gazeuse couplée à la Spectrométrie de Masse (GC-HRMS, GC/MS/MS et GCxGC-
607 TOMS) Principes, applications et perspectives. *Chimie Nouvelle* 93: 72-83. Corpus ID: 42764339

608 Fernandez M.D, Cagigal E, Vega MM, Urzelai A, Babin M, Pro J, Tarazona JV (2005) Ecological risk
609 assessment of contaminated soils through direct toxicity assessment. *Ecotoxicol Environ Saf* 62(2):174-
610 184. <https://doi.org/10.1016/j.ecoenv.2004.11.013>

611 Fernández San Juan MR, Lavariás SML, Aparicio V, Larsen KE, Colman Lerner JE, Cortelezzi A (2023)
612 Ecological risk assessment of pesticides in sediments of Pampean streams, Argentina. *Chemosphere* 313:
613 137598. <https://doi.org/10.1016/j.chemosphere.2022.137598>

614 Ferrans L, Schmieder F, Mugwira R, Marques M, Hogland W (2022). Dredged sediments as plant-
615 growing substrate: estimation of health risk index. *Sci Tot Environ* 846: 157463.
616 <http://dx.doi.org/10.1016/j.scitotenv.2022.157463>

617 Fuentes A, Lloréns M, Saez J, Aguilar MI, Ortuno J, Meseguer V (2004) Phytotoxicity and heavy metals
618 speciation of stabilised sewage sludges. *J Hazard Mater* 108: 161-169.
619 <https://doi.org/10.1016/j.ecoenv.2004.11.013>.

620 Garcia M, Römbke, J, Torres, M, Scheffczyk, A (2008) Effects of three pesticides on the avoidance
621 behavior of earthworms in laboratory tests performed under temperate and tropical conditions. *Environ*
622 *Poll* 153: 450-456. <https://doi.org/10.1016/j.envpol.2007.08.007>

623 Gascón Díez E, Corella JP, Adatte T, Thevenon F, Loizeau J-L (2017) High-resolution reconstruction of
624 the 20th century history of trace metals, major elements, and organic matter in sediments in a
625 contaminated area of Lake Geneva, Switzerland. *Appl Geochem* 78: 1–11. <https://doi.org/10.1016/j.apgeochem.2016.12.007>

627 Gonzalez-Merchan C, Perrodin Y, Sébastien C, Bazin C, Winiarski T, Barraud S (2014) Ecotoxicological
628 characterization of sediments from five stormwater retention basins. *Water Sci Technol* 69(5):1045-51.
629 <https://doi.org/10.2166/wst.2014.006>

630 Gosset A, Polomé P, Perrodin Y (2020) Ecotoxicological risk assessment of micropollutants from treated
631 urban wastewater effluents for watercourses at a territorial scale: Application and comparison of two
632 approaches. *Int J Hyg Environ Health* 224: 113437. <https://doi.org/10.1016/j.ijheh.2019.113437>

633 Greenslade P, Vaughan GT (2003) A comparison of collembola species for toxicity testing of Australian
634 soils. *Pedobiologia* 47: 171-179. <https://doi.org/10.1078/0031-4056-00180>

635 Hankard PK, Bundy JG, Spurgeon DJ, Weeks JM, Wright J, Weinberg C, Svendsen C (2005) Establishing
636 principal soil quality parameters influencing earthworms in urban soils using bioassays. *Environ Poll* 133:
637 199-211. <https://doi.org/10.1016/j.scitotenv.2022.153301>

638 Heise S, Babut M, Casado C, Feiler U, Ferrari BJD, Marziali L (2020) Ecotoxicological testing of
639 sediments and dredged material: an overlooked opportunity? J Soils Sediments 20, 4218–4228.
640 <https://doi.org/10.1007/s11368-020-02798-7>

641 Heupel K (2002) Avoidance response of different Collembolan species to Betanal. Eur J Soil Biol 38:
642 273-276. [https://doi.org/10.1016/S1164-5563\(02\)01158-5](https://doi.org/10.1016/S1164-5563(02)01158-5)

643 How CM, Kuo Y-H, Huang M-L, Liao VH-C (2023) Assessing the ecological risk and ecotoxicity of the
644 microbially mediated restoration of heavy metal-contaminated river sediment. Sci Tot Environ 858,
645 159732. <https://doi.org/10.1016/j.scitotenv.2022.159732>

646 Hund-Rinke K., Lindemann M., Simon M. (2005). Experiences with novel approaches in earthworms
647 testing alternatives. J Soils Sediments 5: 233-239. <https://doi.org/10.1065/jss2005.06.142>

648 ISO 11268-1 (1993). Soil quality-effects of pollutants on earthworms (*Eisenia fetida*). 1. Determination of
649 acute toxicity using artificial soil substrates [S]. International organization for standardization, Geneva.

650 ISO 11268-2 (1998). Soil quality-effects of pollutants on earthworms (*Eisenia fetida*). 2. Determination of
651 effect son reproduction[S]. International organization for standardization, Geneva.

652 JORF (2006). Arrêté du 9 aout 2006 relatif aux niveaux à prendre en compte lors d'une analyse de rejets
653 dans les eaux de surface ou de sédiments marins, estuariens ou extraits de cours d'eau ou canaux relevant
654 respectivement des rubriques 2.2.3.0, 4.1.3.0 et 3.2.1.0 de la nomenclature annexée au décret n° 93-743
655 du 29 mars 1993. Journal Officiel de la République Française 222 : 14082.
656 <https://www.legifrance.gouv.fr/loda/id/JORFTEXT000000423497/>

657 Javorska H, Tlustos P, Kaliszova R (2009) Degradation of Polychlorinated Biphenyls in the Rhizosphere
658 of Rape, *Brassica napus* L. Bull Environ Contam Toxicol, 82:727–731. <https://doi.org/10.1007/s00128-009-9691-y>

660 King RF, Royle A, Putwain PD, Dickinson NM (2006) Changing contaminant mobility in a dredged canal
661 sediment during a three-year phytoremediation trial. Environ Poll 143: 318-326.
662 <https://doi.org/10.1016/j.envpol.2005.11.024>.

663 Kucharski D, Nałęcz-Jawecki G, Drzewicz P, Skowronek A, Mianowicz K, Strzelecka A, Joanna
664 Giebułtowicz J (2022) The assessment of environmental risk related to the occurrence of pharmaceuticals
665 in bottom sediments of the Odra River estuary (SW Baltic Sea). Sci Tot Environ 828, 154446.
666 <https://doi.org/10.1016/j.scitotenv.2022.154446>

667 Kortenkamp A, Faust M, Backhaus T, Altenburger R, Scholze M, Müller C, Ermler S, Posthuma L, Brack
668 W (2019) Mixture risks threaten water quality: the European Collaborative Project SOLUTIONS
669 recommends changes to the WFD and better coordination across all pieces of European chemicals
670 legislation to improve protection from exposure of the aquatic environment to multiple pollutants. Environ
671 Sci Eur 31, 69. <https://doi.org/10.1186/s12302-019-0245-6>.

672 Lamouroux N, Gore JA, Lepori F, Statzner B (2015) The ecological restoration of large rivers needs
673 science-based tools meeting public expectations: an overview of the Rhône project. Freshw Biol 60(6):
674 1069-1084. <https://doi.org/10.1111/fwb.12553>

675 Lancelleur L, Schäfer J, Chiffolleau J-F, Blanc G, Auger D, Renault S, Baudrimont M, Audry S (2011)
676 Long-term records of cadmium and silver contamination in sediments and oysters from the Gironde
677 fluvial–estuarine continuum – Evidence of changing silver sources. Chemosphere 85, 1299–1305.
678 <https://doi.org/10.1016/j.chemosphere.2011.07.036>.

679 Larrose A, Coynel A, Schäfer J, Blanc G, Massé L, Maneux E (2010) Assessing the current state of the
680 Gironde Estuary by mapping priority contaminant distribution and risk potential in surface sediment. Appl
681 Geochem 25, 1912–1923. <https://doi.org/10.1016/j.apgeochem.2010.10.007>.

682 Lasserre J-P, Fack F, Revets, D, Planchon, S Renaut J, Hoffmann L, Gutleb AC, Muller CP, Bohn T
683 (2009) Effects of the endocrine disruptors' atrazine and PCB 153 on the protein expression of MCF-7
684 human cells. J Proteome Res 8(12): 5485–96. <https://doi.org/10.1021/pr900480f>.

685 Lecomte T, Mamindy-Pajany Y, Lors C, Abriak N-E, Bazin C, Vernus E (2019) A methodological
686 approach for ecotoxicological characterization of non-hazardous sediments for their beneficial reuse. J
687 Soils Sediments 20: 2608-2618. <https://doi.org/10.1007/s11368-019-02543-9>.

688 Lécivain N, Aurenche V, Cottin N, Frossard V, Clément B (2018) Multi-contamination (heavy metals,
689 polychlorinated biphenyls and polycyclic aromatic hydrocarbons) of littoral sediments and the associated
690 ecological risk assessment in a large lake in France (Lake Bourget). Sci Tot Environ 619-620: 854-865.
691 <https://doi.org/10.1016/j.scitotenv.2017.11.151>.

692 Liber Y., Mourier B, Marchand P, Bichon E, Perrodin Y, Bedell J-P (2019) Past and recent state of
693 sediment contamination by persistent organic pollutants (POPs) in the Rhône River: Overview of
694 ecotoxicological implications, Sci Tot Environ 646, 1037-1046. [https://doi.org/](https://doi.org/10.1016/j.scitotenv.2018.07.340)
695 [0.1016/j.scitotenv.2018.07.340](https://doi.org/10.1016/j.scitotenv.2018.07.340).

696 Lopes C, Persat H, Babut M (2012) Transfer of PCBs from bottom sediment to freshwater river fish: A
697 food-web modelling approach in the Rhône River (France) in support of sediment management,
698 Ecotoxicol Environment Saf 81: 17–26. <https://doi.org/10.1016/j.ecoenv.2012.04.007>.

699 Luo W, Verweij RA, van Gestel CAM (2014) Determining the bioavailability and toxicity of lead
700 contamination to earthworms requires using a combination of physicochemical and biological methods.
701 Environ Poll 185: 1-9. <https://doi.org/10.1016/j.envpol.2013.10.017>.

702 MacDonald DD, Ingersoll CG, Berger TA (2000a) Development and Evaluation of Consensus-Based
703 Sediment Quality Guidelines for Freshwater Ecosystems. Arch Environ Contam Toxicol 39: 20–31.
704 <https://doi.org/10.1007/s002440010075>

705 MacDonald DD, Dipinto LM, Field J, Ingersoll CG, Lvong ER, Swartz RC (2000b) Development and
706 evaluation of consensus-based sediment effect concentrations for polychlorinated biphenyls. Environ Tox
707 Chem 19: 1403–1413. <https://doi.org/10.1002/etc.5620190524>

708 Maleri R, Reinecke AJ, Reinecke SA (2007). A comparison of nickel toxicity to pre-exposed earthworms
709 (*Eisenia fetida*, *Oligochaeta*) in two different test substrates. Soil Biol Biochem 39(11): 2849–2853.
710 <https://doi.org/10.1016/j.soilbio.2007.06.008>

711 Mamindy-Pajany Y, Hamer B, Roméo M, Gélet F, Galgani F, Durmisi E, Hurel C, Marmier N (2011) The
712 toxicity of composted sediments from Mediterranean ports evaluated by several bioassays. Chemosphere
713 82: 362-369. <https://doi.org/10.1016/j.chemosphere.2010.10.005>

714 Marmin S, Dauvin J-C, Lesueur P (2014) Collaborative approach for the management of harbour-dredged
715 sediment in the Bay of Seine (France). Ocean Coast Manag 102(A): 328-339.
716 <https://doi.org/10.1016/j.ocecoaman.2014.10.012>

717 Martinez-Haro M, Acevedo P, Pais-Costa AJ, Neto JM, Vieira LR, Ospina-Alvarez N, Taggart MA,
718 Guilhermino L, Ribeiro R, Marques JC (2022) Ecotoxicological tools in support of the aims of the
719 European Water Framework Directive: A step towards a more holistic ecosystem-based approach. Ecol
720 Indic 145: 109645. <https://doi.org/10.1016/j.ecolind.2022.109645>

721 Mourier B, Desmet M, Van Metre PC, Mahler BJ, Perrodin Y, Roux G, Bedell J-P, Lefèvre I, Babut M
722 (2014) Historical records, sources, and spatial trends of PCBs along the Rhône River (France). *Sci Tot*
723 *Environ* 476–477 : 568–576. <http://dx.doi.org/10.1016/j.scitotenv.2014.01.026>.

724 Mubedi JL, Devarajan N, Le Faucheur S, Mputu JK, Atibu EK, Sivalingam P, Prabakar K, Mpiana PT,
725 Wildi W, Poté J (2013) Effects of untreated hospital effluents on the accumulation of toxic metals in
726 sediments of receiving system under tropical conditions: Case of South India and Democratic Republic of
727 Congo. *Chemosphere* 93 (6): 1070-1076. <https://doi.org/10.1016/j.chemosphere.2013.05.080>

728 Nicolaus EM, Law RJ, Wright SR, Lyons BP (2015) Spatial and temporal analysis of the risks posed by
729 polycyclic aromatic hydrocarbon, polychlorinated biphenyl and metal contaminants in sediments in UK
730 estuaries and coastal waters. *Mar Pollut Bull* 95 (1): 469-479.
731 <http://dx.doi.org/10.1016/j.marpolbul.2015.03.012>

732 OECD (2010). Test No. 317: Bioaccumulation in Terrestrial Oligochaetes, OECD Guidelines for the
733 Testing of Chemicals, Section 3, OECD Publishing. DOI: 10.1787/9789264090934-en

734 Oleszczuk P (2008) Phytotoxicity of municipal sewage sludge compost related to physico-chemical
735 properties, PAHs and heavy metals. *Ecotoxicol Environ Saf* 69: 496-505.
736 <https://doi.org/10.1016/j.ecoenv.2007.04.006>

737 Oleszczuk P, Rycaj M, Lehmann J, Cornelissen G (2012). Influence of activated carbon and biochar on
738 phytotoxicity of air-dried sewage sludges to *Lepidium sativum*. *Ecotoxicol Environ Saf* 80: 321-326.
739 <https://doi.org/10.1016/j.ecoenv.2012.03.015>

740 Olivier J-M (2016) La restauration du fleuve Rhône, l'ambition d'une reconquête environnementale.
741 RhoneEco. In : Le suivi scientifique de l'a restauration hydraulique et écologique du Rhône (2000-2015).
742 GRAIE, (Groupe de Recherche Rhône-Alpes sur les Infrastructures et l'Eau) Lyon: 4-5.
743 https://www.graie.org/graie/graiedoc/doc_telech/brochure_RhonEco_restoration_ecologique.pdf

744 Palma P, Ledo L, Soares S, Barbosa I R, Alvarenga P (2014) Integrated environmental assessment of
745 freshwater sediments: a chemical and ecotoxicological approach at the Alqueva reservoir. *Environ*
746 *Geochem Health* 36: 209–223. <https://doi.org/10.1007/s10653-013-9559-2>

747 Pesce S, Campiche S, Casado-Martinez C, Ahmed AM, Bonnineau C, Dabrin A, Lyautey E, Ferrari BJD
748 (2020) Towards simple tools to assess functional effects of contaminants on natural microbial and
749 invertebrate sediment communities. *Environ Sci Pollut Res* 27, 6680–6689.
750 <https://doi.org/10.1007/s11356-019-07331-z>

751 Perrodin Y, Babut M, Bedell J-P, Bray M, Clement B, Delolme C, Devaux A, Durrieu C, Garric J,
752 Montuelle B (2006) Assessment of ecological risks related to depositing dredged materials from canals in
753 northern France on soil. *Environ Int* 32: 804-814. <https://doi.org/10.1016/j.envint.2006.05.003>.

754 Perrodin Y, Bazin C, Bony S, Devaux A, Bertrand-Krajewski J-L, Cren-Olivé C, Brelot E (2013) A priori
755 assessment of ecotoxicological risks linked to building a hospital. *Chemosphere* 90: 1037-1046.
756 <https://doi.org/10.1016/j.chemosphere.2012.08.049>.

757 Perrodin Y, Donguy G, Bazin C, Volatier L, Durrieu C, Bony S, Devaux A, Abdelghafour M, Moretto R
758 (2012) Ecotoxicological risk assessment linked to infilling quarries with treated dredged seaport
759 sediments. *Sci Total Environ* 431: 375–384. <https://doi.org/10.1016/j.scitotenv.2012.05.069>.

760 Phytotoxkit (2004). Seed germination and early growth microbio-test with higher plants. Standard
761 operation procedure. MicroBioTests Inc, Nazareth, Belgium, pp. 1-24.

762 Piégay H , Radakovitch O, Arnaud F, Belletti B, Camenen B, Cassel M, Coquery M, Dabrin A, Depret T,
763 Fressard M, Ghaffarian H, Gruat A, Janssen P, Lang M, Delile H, Le Coz J, Lepage H, Lucas M, Mourier
764 B, Paireaud I, Seignemartin G, Tal M, Serlet A, Tena A, Vazquez- Tarrío D, Winiarski T, Morandi B,

765 Chateauminois A, Clémens A (2022) L'Observatoire des Sédiments du Rhône. 12 années de recherche
766 pour la connaissance et la gestion hydro-sédimentaire du fleuve. Bilans et perspectives scientifiques.
767 GRAIE, Lyon. URL: <https://hal-cnrs.archives-ouvertes.fr/TP/hal-03869998v1>

768 Pye K, Blott SJ (2004) Particle size analysis of sediments, soils and related particulate materials for
769 forensic purposes using laser granulometry. *Forensic Sci Int* 144(1):19-27.
770 <https://doi.org/10.1016/j.forsciint.2004.02.028>.

771 Schaefer M (2004) Assessing 2,4,6-trinitrotoluene (TNT)-contaminated soil using three different
772 earthworm test methods. *Ecotoxicol Environ Saf* 57: 74-80. <https://doi.org/10.1016/j.ecoenv.2003.08.005>

773 Sethy SK, Ghosh S (2013) Effect of heavy metals on germination of seeds. *J Nat Sci Biol Med* 4(2): 272–
774 275. <https://doi.org/10.4103/0976-9668.116964>

775 Soetan O, Nie J, Feng H (2022). Preliminary environmental assessment of metal-contaminated sediment
776 dredging in an Urban river, New Jersey, USA. *Mar Pollut Bull* 184: 114212.
777 <https://doi.org/10.1016/j.marpolbul.2022.114212>

778 Sousa A, Pereira R, Antunes SC, Cachada A, Pereira E, Duarte AC, Gonçalves F (2008) Validation of
779 avoidance assays for the screening assessment of soils under different anthropogenic disturbances.
780 *Ecotoxicol Environ Saf* 71 (3): 661-670. <https://doi.org/10.1016/j.ecoenv.2008.02.004>.

781 Spurgeon DJ, Hopkin SP (1996) Effects of variations of the organic matter content and pH of soils
782 on the availability and toxicity of zinc to the earthworm *Eisenia fetida*. *Pedobiologia* 40: 80–96.
783 <http://dx.doi.org/10.1007/s001289900332>

784 Svensson N, Norén A, Modin O, Fedje KK, Rauch S, Strömvall A-M, Sköld YA (2022) Integrated cost
785 and environmental impact assessment of management options for dredged sediment. *Waste management*
786 138:30-40. <https://doi.org/10.1016/j.wasman.2021.11.031>

787 Tato L, Tremolada P, Ballabio C, Guazzoni N, Parolini M, Caccianiga M, Binelli A (2011) Seasonal and
788 spatial variability of polychlorinated biphenyls (PCBs) in vegetation and cow milk from a high-altitude
789 pasture in the Italian Alps. *Environ Pollut* 159(10):2656-64. <https://doi.org/10.1016/j.envpol.2011.05.035>.

790 Thorel M, Piégay H, Barthelemy C, Rapple B, Gruel CR, Marmonier P, Winiarski T, Bedell J-P, Arnaud
791 F, Roux G, Stella JC, Seignemartin G, Tena-Pagan A, Wawrzyniak V, Roux-Michollet D, Oursel B,
792 Fayolle S, Bertrand C, Franquet E (2018) Socio-environmental implications of process-based restoration
793 strategies in large rivers: should we remove novel ecosystems along the Rhône (France)? *Reg Environ*
794 *Change* 18: 2019–2031. <https://doi.org/10.1007/s10113-018-1325-7>.

795 Todaro F, De Gisi S, Labianca C, Notarnicola M (2019) Combined assessment of chemical and
796 ecotoxicological data for the management of contaminated marine sediments. *Environ Eng Manag J* 18,
797 2287-2296.

798 Touzé S, Bataillard P (2011) Caractérisation d'un sédiment de curage contaminé par des PCB. Rapport
799 BRGM /RP 59491-FR, Orléans, France, mai 2011, 40p.

800 Vauclin S, Mourier B, Dendievel A-M, Marchand P, Vénisseau A, Morereau A, Lepage H, Eyrolle F,
801 Winiarski T (2021) Temporal trends of legacy and novel brominated flame retardants in sediments along
802 the Rhône River corridor in France. *Chemosphere* 271: 129889.
803 <https://doi.org/10.1016/j.chemosphere.2021.129889>.

804 Volatier L, Clément B, Devaux A, Durrieu C, Bedell J-P, Perrodin Y (2009) Benefits and limits of
805 sediment toxicity tests as an aid to decision making. *Environ Technol* 30(8):865-70.
806 <https://doi.org/10.1080/09593330902990170>.

807 Wang F, Leung AOW, Wu SC, Yang MS, Wong MH (2009) Chemical and ecotoxicological analyses of
808 sediments and elutriates of contaminated rivers due to e-waste recycling activities using a diverse battery
809 of bioassays. *Environ Poll* 157: 2082-2090. <https://doi.org/10.1016/j.envpol.2009.02.015>.

810 White J C (2001) Plant-facilitated mobilization and translocation of weathered 2, 2-bis(p-chlorophenyl)-
811 1,1-dichloroethylene (p,p'-DDE) from an agricultural soil. *Environ Toxicol Chem*;20(9): 2047-52. [https://doi.org/10.1897/1551-5028\(2001\)020<2047:pfmato>2.0.co;2](https://doi.org/10.1897/1551-5028(2001)020<2047:pfmato>2.0.co;2).

813 Whitfield Aslund M L, Zeeb B, Rutter A, Reimer K J (2007) *In situ* phytoextraction of polychlorinated
814 biphenyl – (PCB) contaminated soil. *Sci Tot Environ* 374(1): 1-12.
815 <https://doi.org/10.1016/j.scitotenv.2006.11.052>.

816 Whitfield Aslund M L, Rutter A, Reimer K J, Zeeb B (2008) The effects of repeated planting, planting
817 density, and specific transfer pathways on PCB uptake by *Cucurbita pepo* grown in field conditions. *Sci*
818 *Tot Environ* 405 (1-3): 14-25. <https://doi.org/10.1016/j.scitotenv.2008.07.066>.

819 Whitfield Åslund ML, Lunney A I, Rutter A, Zeeb B (2010) Effects of amendments on the uptake and
820 distribution of DDT in *Cucurbita pepo* ssp *pepo* plants. *Environ Poll* 158: 508–513
821 <https://doi.org/10.1016/j.envpol.2009.08.030>.

822 Whitfield Åslund ML, Stephenson GL, Simpson AJ, Simpson MJ (2013) Comparison of earthworm
823 responses to petroleum hydrocarbon exposure in aged field contaminated soil using traditional ecotoxicity
824 endpoints and ¹H NMR-based metabolomics. *Environ Poll* 182: 263-268.
825 <https://doi.org/10.1016/j.envpol.2013.07.026>.

826 Wierzbicka M, Obidzińska J (1998) The effect of lead on seed imbibition and germination in different
827 plant species. *Plant Science* 137: 155–171. [http://dx.doi.org/10.1016/S0168-9452\(98\)00138-1](http://dx.doi.org/10.1016/S0168-9452(98)00138-1)

828 Zhao, F., McGrath, S. P., Crossland, A. R. (1994). Comparison of three wet digestion methods for the
829 determination of plant sulphur by inductively coupled plasma atomic emission spectroscopy (ICP-AES).
830 *Commun Soil Sci* 25: 407–418. <https://doi.org/10.1080/00103629409369047>.

831 Zhou, S.-P., Duan, C.-Q., Fu, H., Chen, Y.-H., Wang, X.-H., Yu, Z.-F. (2007). Toxicity assessment for
832 chlorpyrifos-contaminated soil with three different earthworms test methods. *J Environ Sci* 19: 854-858.
833 [https://doi.org/10.1016/s1001-0742\(07\)60142-9](https://doi.org/10.1016/s1001-0742(07)60142-9).