



**HAL**  
open science

## **Does pollution influence small mammal diet in the field? A metabarcoding approach in a generalist consumer**

Shinji Ozaki, Clémentine Fritsch, Benoît Valot, Frédéric Mora, Renaud Scheifler,  
Francis Raoul, Thierry Cornier

### ► **To cite this version:**

Shinji Ozaki, Clémentine Fritsch, Benoît Valot, Frédéric Mora, Renaud Scheifler, et al.. Does pollution influence small mammal diet in the field? A metabarcoding approach in a generalist consumer. *Molecular Ecology*, 2018, 27 (18), pp.3700-3713. <10.1111/mec.14823>. <hal-02368039>

**HAL Id: hal-02368039**

**<https://hal.science/hal-02368039v1>**

Submitted on 7 Feb 2022

**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.



HAL Authorization

1 **Does pollution influence small mammal diet in the field? A metabarcoding approach**  
2 **in a generalist consumer.**

3  
4 Shinji Ozaki<sup>\*1</sup>, Clémentine Fritsch<sup>1</sup>, Benoit Valot<sup>1</sup>, Frédéric Mora<sup>2</sup>, Thierry Cornier<sup>3</sup>,  
5 Renaud Scheifler<sup>1#</sup> and Francis Raoul<sup>1#</sup>

6  
7 <sup>1</sup> Laboratoire Chrono-environnement, UMR CNRS 6249 UoC INRA, Université  
8 Bourgogne Franche-Comté, 16 route de Gray, 25030 Besançon cedex, France

9 <sup>2</sup> Conservatoire Botanique National de Franche-Comté, Observatoire Régional des  
10 Invertébrés, 7 rue Voirin, 25000 Besançon, France

11 <sup>3</sup> Centre régional de phytosociologie agréé Conservatoire Botanique National de Bailleul,  
12 Hameau de Haendries, F-59270 Bailleul, France

13  
14 \* Corresponding author: Shinji Ozaki

15 Phone number: +33 (0)3 81 66 65 98

16 E-mail address: shinji.ozaki@univ-fcomte.fr

17 # both authors contributed equally to supervising this work

18

19 **Abstract:**

20 Mammals are mainly exposed to trace metals via consuming contaminated food. Several  
21 studies have demonstrated relationships between metal concentrations in food and in animal  
22 tissues. However, potential effects of trace metals on feeding behaviour of wildlife have been  
23 poorly documented under field conditions, despite experimental evidence showing that food  
24 selection is impacted by resource contamination. Here we test the hypothesis that the diet of  
25 a generalist rodent, the wood mouse (*Apodemus sylvaticus*), is altered by soil trace metal  
26 contamination in the field. Wood mice were sampled in spring and in autumn along a  
27 gradient of soil contamination in the surroundings of a former smelter located in northern  
28 France. Available resources in the field were inventoried and the diet of the animals was  
29 analysed using DNA “metabarcoding”. We demonstrated that (i) relationship between the  
30 resource richness in the diet and their richness in the field was altered by soil metal  
31 contamination. Wood mice specialised their diet along the gradient of soil metal  
32 contamination for both plant and invertebrate resources in spring. We also showed that (ii)  
33 preference for Salicaceae, a plant family accumulating metals, decreased when soil  
34 contamination increased. These results suggest that environmental trace metal pollution  
35 could act as a force modulating trophic interactions in terrestrial food webs, thereby affecting  
36 wildlife exposure to contaminants by trophic route.

37

38 **Keywords:**

39 Biodiversity; Diet diversity; Food selection; Trace metals; *Apodemus sylvaticus*

40

41 **Introduction**

42 Feeding behaviour is determined by life history traits and life stages of animals (Schoener,  
43 1971; Pyke et al., 1977), abundance and distribution of food resources in nature (White,  
44 2008; Crampton et al., 2011), and food quality (i.e. nutrient and toxin content) (Butet, 1986a;  
45 Kerley & Erasmus 1991; Schmidt, 2000; Lewis et al., 2001). All these factors are driven by  
46 ecological determinants like season, climate, community interactions or landscape features  
47 (Polis et al., 1997; Visser & Both 2005; White, 2008). The optimal foraging theory assumes  
48 that animals consume resources providing high energy intake per unit time (Stephens &  
49 Krebs, 1986) and that the diet of animals tends to be specialized when food density is high  
50 (MacArthur & Pianka, 1966). However, animals require not only energy but also nutrients  
51 like vitamins or essential metals. The nutrient requirements can thus hardly be satisfied from  
52 a single plant species and thus animals seek a variety of diet for nutrient balance (i.e. nutrient  
53 balance hypothesis) (Westoby, 1978). Likewise, Freeland and Janzen (1974) proposed that  
54 generalist mammalian herbivores are unable to detoxify large amounts of similar plant  
55 secondary metabolites (i.e. those from a single plant species) and thus a variety of plant  
56 sources allow them to avoid similar plant toxin load (i.e. detoxification limitation  
57 hypothesis). When the use of a resource is disproportional to its availability in the field, the  
58 feeding process by which an animal chooses the resource is called “selection” (Johnson,  
59 1980; Manly et al., 2002). The nature of selection (i.e. preference or avoidance) reflects the  
60 likelihood of a given resource for being chosen when this resource is offered with others  
61 (Johnson, 1980). The food selection hence manifests a relationship between individual  
62 requirements and resource constraints.

63 Trace metals (TM) are environmental pollutants of concern for both human health and  
64 wildlife. They occur often as a legacy of past industrial activities. Even if the emissions of  
65 many of them have been reduced in the past decades thanks to environmental regulation and

66 technical improvements at least in so-called developed countries, the pollution still persists  
67 over vast surfaces due to non-degradability of metals and to their current emissions in many  
68 countries (Kabata-Pendias, 2011; EMEP, 2013). TM pollution may have effects on plants  
69 and invertebrates at several levels of organisation, from sub-individual physiological effects  
70 reducing survival or reproduction to the modification of community composition and  
71 structure (Walker et al., 2012). For instance, a recent meta-analysis about the responses of  
72 terrestrial biota to TM pollution -dealing with 206 pollution point sources worldwide-  
73 showed a decrease of diversity for most of the taxa (producers, secondary consumers and  
74 decomposers) (Kozlov & Zvereva, 2011). In addition, Dazy et al. (2009) demonstrated that  
75 plant species richness correlated negatively to TM concentrations in soils in eastern France.  
76 Metal pollution can also reduce species richness in earthworm communities (Spurgeon &  
77 Hopkin, 1999; Nahmani et al., 2003) and impact the structure and composition of soil macro-  
78 fauna communities (Nahmani & Lavelle, 2002). Trophic transfer , i.e. the transfer of matter  
79 via food consumption, is generally considered to be the main route for TM from the  
80 environment to animals of higher trophic levels (Shore & Rattner, 2001; Smith et al., 2007).  
81 For small mammals, several studies have demonstrated relationships between soil pollution  
82 by TM, contamination of food, and subsequently, high concentrations of TM in wild animals'  
83 tissues (Hunter & Johnson, 1982; Hunter et al., 1987a; Torres & Johnson, 2001; Rogival et  
84 al., 2007; van den Brink et al., 2010). Avoidance of TM contaminated food can be a  
85 mechanism that mitigates organisms' exposure to contaminants (Gillet & Ponge, 2003). This  
86 has been reported for the wood mouse (*Apodemus sylvaticus*) under laboratory settings but  
87 not confirmed in natural conditions (Beernaert et al., 2008). To the best of our knowledge,  
88 no study has reported variations of richness in the diet and selection as a response to both  
89 the variation of resource availability and the contamination of the environment *in natura* in  
90 terrestrial ecosystems.

91 One of the main difficulties in assessing wildlife diet is the identification of items actually  
92 consumed. Identifying items with the highest taxonomic resolution is almost impossible with  
93 classical macro- or microscopy-based methods, which then makes it difficult to match diet  
94 with field records of resources. Recently, DNA-based techniques have arisen as a powerful  
95 approach for the taxonomic identification. Although DNA-based identification is not novel,  
96 current technological developments such as DNA metabarcoding approach enable to  
97 simultaneously identify different taxa in environmental samples, like soils or faeces,  
98 containing multiple specimens and potentially highly degraded DNA (reviewed in Valentini  
99 et al., 2009a; Taberlet et al., 2012; Pompanon et al., 2012). The efficiency of this approach  
100 for analysing the diet has been demonstrated in various species including herbivorous  
101 mammals, birds and invertebrates (Soininen et al., 2009; Valentini et al., 2009b),  
102 insectivorous bats (Bohmann et al., 2011), or omnivorous animals like the brown bear (De  
103 Barba et al., 2013).

104 In the present study, we focused on the wood mouse (*Apodemus sylvaticus*), which is a  
105 generalist small mammal (Watts, 1968; Hansson, 1985; Butet, 1990) and therefore a model  
106 of choice when studying diet plasticity as a response to environmental variations. Indeed,  
107 provided its opportunistic feeding behaviour, the diet richness (i.e. total number of resources'  
108 species in the diet) of the wood mouse is expected to be positively correlated with resource  
109 richness in the field. We hypothesised that: (i) diet richness of the wood mouse would be  
110 negatively correlated to TMs contamination of soils given the reported negative effects of  
111 pollution on plant and invertebrate richness in community, and (ii) food selection by the  
112 wood mouse would be affected by TM contamination in soils possibly due to avoidance  
113 behaviour.

114

115 **Materials and methods**

116 **Study sites**

117 This study was carried out in the surroundings of the former lead (Pb) and zinc (Zn) smelter  
118 named “Metaleurop Nord” located in northern France (Noyelles Godault, Hauts-de-France,  
119 France). The surrounding soils were highly contaminated by TM, mainly cadmium (Cd), Pb,  
120 and Zn, over a large area of about 120 km<sup>2</sup> because of metal-contaminated dust released for  
121 decades from this smelter and another one named “Umicore”, located about 4 km east of the  
122 first (Sterckeman et al., 2002; Douay et al., 2008; Douay et al., 2009; Waterlot et al., 2016).  
123 A study area of 40 km<sup>2</sup> was defined and divided into 160 sites of 25 ha (500m x 500m)  
124 (Fritsch et al., 2010), from which seven are used in the present study (see below). Total TM  
125 concentrations in the soil of woody patches ranged from 0.1 to 236.5 mg kg<sup>-1</sup> of dry soil for  
126 Cd, from 16 to 7331 mg kg<sup>-1</sup> of dry soil for Pb, and from 44 to 7264 mg kg<sup>-1</sup> of dry soil for  
127 Zn in an area of 40 km<sup>2</sup> in the surroundings of the former smelter (Fritsch et al., 2010). These  
128 values largely exceed background TM concentrations in soils of the region Hauts-de-France  
129 (Sterckeman et al., 2007). Those TM concentrations in woody patches were used in our  
130 modelling (cf. below). The seven sites are located along a soil pollution gradient and  
131 composed of three types of dominant habitats (woodland, urban features, arable lands) to  
132 provide a wide range of resource richness in the field (for details see Supporting Information  
133 SII).

134

135 **Rodent trapping**

136 At each site, we trapped wood mice in spring (April) and in autumn (September and October)  
137 2012 according to Fritsch et al. (2011) and in accordance with current French legislation  
138 about ethics and use of animals in research. Briefly, small break-back traps were used with  
139 peanut as rodent’s bait. Ten lines, composed of 10 traps 3 m spaced each, were used per site  
140 and per season. All trap lines were set in woody habitats of the sites and their location was

141 geo-referenced. The trap lines were checked in the morning for three consecutive days and  
142 re-set and/or re-baited, if necessary. After being weighed in the field, animals were  
143 immediately frozen at -20°C for further analyses.

144 Two hundred and forty-six rodents (117 in spring and 129 in autumn; 116 females and 127  
145 males) were used in the study. Age was estimated using body weight according to four  
146 classes (approximately, class I < 50 days < class II < 130 days < class III < 230 days < class  
147 IV), referring to Vandorpe & Verhagen (1980) and Tête et al. (2014). Age structure differed  
148 between seasons: no mouse of age class I was found in spring, and the proportions of animals  
149 in the other 3 classes were significantly different ( $\chi^2 = 31.04$ , p-value < 0.001) such that  
150 animals in spring were older than in autumn.

151

#### 152 **Stomach content samples**

153 Consumed items were identified by biomolecular analysis of stomach contents (SC). Wood  
154 mice were thawed at room temperature. Their SC was extracted from the body with a spatula,  
155 which was thoroughly cleaned with disposable tissue, then washed with ultra-pure water  
156 (18.2 M $\Omega$ /cm<sup>2</sup>) and wiped off with other tissue between each extraction. Remaining bait was  
157 removed from SC. Each SC was then homogenized and split into two aliquots: one (about  
158 10 mg) was stored in 95% ethanol for metabarcoding analysis (see below), the other was  
159 reserved for another study. For 13 mice among the 246 rodents, faeces taken from intestines  
160 were used as an alternative material for food identification because their stomach was empty.  
161 The 13 mice were equitably distributed according to the explanatory factors tested in this  
162 study.

163

#### 164 **Metabarcoding analysis**

165 The DNA extraction, amplification and the PCR purification were performed at SPYGEN

166 facilities (www.spygen.com). DNeasy Blood and Tissue Kit (Qiagen GmbH) was used for  
167 extracting total DNA, following the manufacturer's instruction. DNA amplification was  
168 performed with three sets of primers (Table 1): the P6 loop of the chloroplast *trnL* (UAA)  
169 intron *g/h* (Taberlet et al., 2007) was used for identifying general plant species; primer  
170 targeting a short fragment of mitochondrial 16S gene (16S mtDNA) was used for arthropods  
171 and molluscs DNA (*16SMAV-F/16SMAV-R*) (De Barba et al., 2013); and primer targeting  
172 short region of 16S mtDNA for earthworms (*ewD/ewE*) (Bienert et al., 2012). PCR  
173 amplifications were carried out on Applied Biosystems Veriti 96 Wells (Life Technologies).  
174 The amplification was realized in a final volume of 25  $\mu$ L using 3  $\mu$ L of DNA extract. Two  
175 PCR replicates were performed per each sample. For the amplification of arthropod and  
176 mollusc DNA, 2  $\mu$ M of a blocking primer for mammal's DNA (*MamMAVBI*; De Barba et  
177 al., 2013) was added in the PCR mix. The amplification mixture contained 1 U of AmpliTaq  
178 Gold DNA Polymerase (Applied Biosystems), 10 mM of Tris-HCl, 50 mM of KCl, 2.5 mM  
179 of MgCl<sub>2</sub>, 0.2 mM of each dNTP, 0.2  $\mu$ M of group-specific primers, 0.2  $\mu$ g/ $\mu$ L of bovine  
180 serum albumin (BSA, Roche Diagnostic) and ultra-pure water (18.2 M $\Omega$  /cm<sup>2</sup>) to bring each  
181 sample to the final volume. The mixture was denatured at 95°C for 10 min, followed by 45  
182 cycles of 30 s at 95°C, 30 s at 50°C for *trnL-g/h*, at 55°C for *16SMAV-F/16SMAV-R* and at  
183 58°C for *ewD/ewE* and 1 min at 72°C, followed by a final elongation at 72°C for 7 min.  
184 Extraction and PCR negative controls were analysed in parallel in order to monitor potential  
185 contamination. After amplification, the samples were titrated using capillary electrophoresis  
186 (QIAxcel; Qiagen GmbH) and purified using a MinElute PCR purification kit (Qiagen  
187 GmbH). Before sequencing, purified DNA was titrated again using capillary electrophoresis.  
188 The purified PCR products were pooled in equal volumes, to achieve an expected sequencing  
189 depth of 10,000 reads per sample. Libraries were prepared using TruSeq Nano DNA genomic  
190 kit (Illumina) and a pair-end sequencing (2x100 bp) was carried out with an Illumina HiSeq

191 sequencer (Illumina) using TruSeq SBS Kit v3 (Illumina) following the manufacturer's  
192 instructions. Library preparation and sequencing were performed at Fasteris facilities  
193 (www.fasteris.com).

194 Reads were handled by the software Mothur pipeline (Schloss et al., 2009). Briefly, forward  
195 and reverse reads were assembled in contig sequences. Then, sequences were filtered based  
196 on length (20-90bp for *trnL-g/h*; 36-38bp for *16SMAV*; 69-81bp for *ewD/ewE*), homo-  
197 polymer (less than 10 nucleotides) and no ambiguous nucleotides. After de-replication  
198 (occurrence count of each different sequence), only unique sequences with a minimum count  
199 of 10 (sum of all samples) were used for further analysis.

200

#### 201 **Field inventory of plant and invertebrate resources**

202 Vegetation survey was realized once per site from 4<sup>th</sup> June to 5<sup>th</sup> September 2012. Three  
203 different strata were defined based on height: tree stratum (woody species > 8 m high), shrub  
204 stratum (woody species < 8 m high), and herbaceous stratum. Taxa were identified at species  
205 level in the field. Nomenclature of species followed Lambinon et al. (2004) and Dudman &  
206 Richards (1997). Cover-abundance of vascular plant species was visually estimated as the  
207 vertically projected area following Braun-Blanquet et al. (1952). Vegetation habitats were  
208 determined by plants' composition of one or more strata and delineated as polygons with the  
209 aid of aerial pictures. The polygons were geo-referenced and digitalized using the QGIS  
210 software (<http://www.qgis.org/>). We listed 236 different plant taxa in the field (see SI1 for  
211 the taxonomic richness in the field per site and SI2 for the list of the taxa and their abundance  
212 per site).

213 Invertebrate fauna was sampled at the same time as the rodent trapping (i.e. in the two  
214 seasons). Ground-dwelling invertebrates were captured near rodent traps using three pitfall  
215 traps per line. Flying invertebrates were captured with yellow pan traps containing a mix of

216 soap and water. Four yellow pan traps were set in each site. Yellow colour attracts especially  
217 flower visiting insects. The traps were checked every morning for three consecutive days.  
218 All traps were geo-referenced. Captured invertebrates were conserved in ethanol or in freezer  
219 at -20°C and identified to the highest possible taxonomic resolution by morphological  
220 characteristics (the main references are Coulon, 2003; Forel & Leplat, 2001; Jeannel, 1941;  
221 Trautner & Geigenmueller, 1987). Captured ground-dwelling and flying invertebrates  
222 consisted of 88 and 95 taxa, respectively (see SI1 for the taxonomic richness in the field per  
223 site and SI2 for the list of the taxa and their abundance per site). Because of the different  
224 taxonomic resolution, we used mainly family level (and higher levels) for invertebrate  
225 richness in the field (cf below): 24 and 95 families (or higher) in ground-dwelling and flying  
226 invertebrates, respectively.

227 We then estimated available resources for wood mice near their trapping location. Although  
228 notion of “availability” is distinct from “abundance” in selection studies (Manly et al., 2002),  
229 we considered them equivalent for wood mice given its generalist diet behaviour. Plant  
230 resource availability was estimated by the cover-abundance of each plant species found  
231 within a buffer of 10 m around trap lines (referred to as “buffer” thereafter). This buffer is  
232 equivalent to an area of about 1000 m<sup>2</sup>, considered as the average size of wood mouse vital  
233 domain (Quéré & Le Louarn, 2011). We considered the data set collected between June and  
234 September to be representative of food availability in spring and autumn. Since the traps  
235 used to capture invertebrates (pitfall trap and yellow pan trap) were inappropriate for  
236 catching earthworms and molluscs, both earthworm and mollusc richness in the field were  
237 unknown. Resource availability of ground-dwelling invertebrates in buffers both in spring  
238 and in autumn was estimated by the abundance of each taxa found per trap line (sum of the  
239 three traps per line), whereas availability of flying invertebrates was based on abundance in  
240 the yellow pan trap nearest to the trap line.

241

## 242 **Diet diversity analyses**

### 243 Sequence data for diet diversity analyses

244 Diet diversity analyses were performed by employing Molecular Operational Taxonomic  
245 Units (MOTUs): groups of DNA sequences clustered according to similarity, which are thus  
246 independent of any reference database (Sun et al., 2012). Sequences were clustered for each  
247 primer with average neighbour algorithm using Needlman-Wunsch distance. Distances  
248 cutting clusters were arbitrarily chosen as 0.032 for sequences obtained from primer *trnL-*  
249 *g/h*, as 0.042 for *16SMAV* and as 0.034 for *ewD/ewE* (see SI3 for sequence data of plant,  
250 arthropod and earthworm MOTUs). Deagle et al. (2013) showed that the number of  
251 sequences obtained from degraded DNA is likely not proportional to the biomass of each  
252 taxa really eaten. The quantitative sequence information data were therefore converted into  
253 presence/absence of MOTUs using an occurrence threshold of 100 to remove background  
254 sequencing data.

255

### 256 Diet richness and modelling

257 Richness in the diet of each mouse was calculated as the total number of MOTUs per mouse  
258 by three types of food: plant, arthropod and mollusc (referred to as “arthropod” thereafter),  
259 or earthworm. We assumed that plants potentially consumed by the invertebrates would not  
260 artificially increase the number of plant MOTUs consumed by the rodent (i.e. there may be  
261 no secondary predation, for detail see Harwood et al., 2001) because the number of plant  
262 MOTUs was not significantly different between SCs containing or not containing  
263 invertebrate food (arthropod and/or earthworm) (data not shown). The diet richness per  
264 mouse was compared among seasons, wood mouse sexes and age-classes, and landscape  
265 features using the nonparametric Wilcoxon-Mann-Whitney (for comparison of two

266 categories) or Kruskal-Wallis tests (for more than two categories), due to skewed distribution  
267 of MOTUs data. Frequency of occurrence was also compared by chi-square goodness-of-fit  
268 test.

269 Relationship between diet richness, richness in the field and TM concentrations in soil was  
270 assessed by statistical modelling. In order to estimate the richness in the diet of all mice in a  
271 buffer, the total number of MOTUs of all mice captured over a given buffer (referred to as  
272 “diet richness” thereafter) was calculated, per type of food or as total diet richness by  
273 summing plant, arthropod and earthworm MOTUs. The relationship was analysed using  
274 multimodel inference on the basis of the information-theoretic approaches (Burnham &  
275 Anderson, 2002). A set of candidate models was established by combinations of several  
276 explanatory variables. Second order Akaike Information Criterion (AICc: corrected AIC for  
277 small sample size in relation to the number of model parameters) was used for ranking the  
278 proposed models. Probability of the “best” model over the set was estimated by Akaike  
279 weight ( $w$ ). In situations where the best model was uncertain (i.e.  $w$  of the first ranked model  
280  $< 0.9$ ), we applied a model averaging approach for reducing model selection uncertainty. A  
281 confidence set of candidate models, whose  $\Delta AICc < 6$  (difference of AICc from the first  
282 ranked model  $< 6$ , which is equivalent to a likelihood ratio from the first ranked model  $>$   
283  $0.05$ ), was re-selected, and each parameter of those re-proposed models was averaged by re-  
284 calculated Akaike weight (Burnham & Anderson, 2002). When a given parameter was not  
285 included in a proposed model, the value of the parameter was considered as 0 in this model  
286 (Burnham & Anderson, 2002; Lukacs et al., 2010).

287 All candidate models were established as generalized linear models (GLM) with logarithm  
288 link function for Poisson distribution. Models were built separately per season, given  
289 potential seasonal diet variation of the animal (Butet, 1986b; Abt & Bock, 1998), as well as  
290 per TM, considering their different effects on organisms and transfer pathways (Eisler, 2000).

291 We used the diet richness in buffer as response variable and number of analysed mice per  
292 trap line as one of the explanatory variables. Soil TM concentrations of woody patch  
293 characterised by Fritsch et al. (2010) were applied to buffer set in or near a given woody  
294 patch. Mice of age class I were discarded from modelling because diet of juveniles could  
295 differ from adults (Watts, 1968; Montgomery & Montgomery, 1990). The modelling was  
296 carried out on 38 buffers in spring and 37 buffers in autumn. Number of mice used in the  
297 analysis was 181 and their number per line ranged from 1 to 6.

298 For plant richness in the diet, combinations of three variables and one interaction were  
299 considered as candidate models: number of mice, TM concentrations in soils of woody  
300 habitat (referred by the data of Fritsch et al., 2010; see "Study site" above) with a natural  
301 logarithm transformation, and plant richness in the field, as well as an interaction between  
302 the TM concentrations and the plant richness in the field (i.e. 10 models including the  
303 intercept only model). For arthropod richness in the diet, four variables and two interactions  
304 were used: mouse number, TM concentrations, both ground-dwelling and flying invertebrate  
305 richness in the field, as well as two interactions between the TM concentration and the  
306 richness in the field (i.e. 26 models). For earthworm richness in the diet, we used only two  
307 variables: mouse number and TM concentrations (i.e. four models). For total diet richness,  
308 we used three variables and one interaction: mouse number, TM concentrations, and total  
309 richness in the field (sum of plant and both ground-dwelling and flying invertebrate richness  
310 at family level), as well as interaction between the TM concentrations and the total richness  
311 in the field (i.e. 10 models). Overdispersion (observed variance higher than theoretical one)  
312 was checked according to Cameron & Trivedi (1990). Explanatory value of our final models  
313 were estimated by deviance  $R^2$  ( $R^2_D$ : proportion of deviance explained by the given model)  
314 (Zuur, 2009).

315

316 **Food selection analyses**

317 Sequence data for selection analysis and association with field survey data

318 Reference sequences of the species recorded in our study area were taken from the GenBank  
319 sequence database (www.ncbi.nlm.nih.gov/genbank). Each sequence extracted from SCs  
320 (i.e., query sequence) was compared to the reference sequences for checking their similarity,  
321 where we accepted difference of only one nucleotide. When query sequences could match  
322 with several reference sequences, they were gathered into one group composed of the  
323 corresponding species, called “sequence group (Grp)” (see SI4 for details of sequence  
324 groups). Non assigned query sequences were excluded. Query sequences with occurrence  
325 lower than 100 were not considered. Food Grp data were finally converted into  
326 presence/absence. We represented Grps using both ID number and family name(s)  
327 corresponding to their component species. We applied such method only to plant food  
328 because almost all invertebrate resources in the field were identified to family level only.

329

330 Plant food selection and modelling

331 Frequency of occurrence of each plant food item (i.e. percentage of number of occurrences  
332 of a given item compared to the total number of occurrence of all items, referring to Klare  
333 et al., 2011) was calculated at family level using Grp. Food selection was assessed, for each  
334 plant food item, per season, comparing the frequency of occurrence and relative proportion  
335 of resources in the field (i.e. cover-abundance of a given plant food of all strata compared to  
336 the total cover-abundance of all plants of all strata). Plant resource taxa in the field were  
337 gathered into corresponding Grp and then into the same plant family. Resource availability  
338 was estimated based on the abundance of the plant families in each buffer, summed up per  
339 family for all mice in each season. Significance of food selection was evaluated per plant  
340 family using Bailey’s confidence intervals at p-value < 0.05 (Bailey, 1980; Cherry, 1996):

341 when relative availability of a given resource was located below or above the confidence  
342 interval of its frequency of occurrence, this resource is considered as preferred or avoided,  
343 respectively.

344 Preference change according to richness in the field and/or TM concentrations in soil was  
345 then assessed by multimodel inference approaches. First, food selection was estimated in  
346 each buffer as described above. Buffers where selection for a given resource was observed  
347 were considered 1 (i.e. selection for the resource is present), whereas the other buffers were  
348 considered 0 (i.e. selection for the resource is absent). We used GLM with logit link function  
349 for assessing change of proportion of buffers where preference was observed. There were  
350 only one plant family (Salicaceae) for which selection was significantly observed in more  
351 than five buffers, allowing statistical analyses. Salicaceae was preferred in spring in six  
352 among 38 buffers. Candidate models were established by combinations of three variables  
353 and one interaction: mice number in buffer, plant richness in the field, and TM concentrations,  
354 as well as an interaction between the richness in the field and the TM concentrations. The  
355 modelling was performed for each TM. Model selection methods were the same as described  
356 above.

357 All statistical analyses were computed using the statistical software R (ver. 3.4.2; R  
358 Development Core Team). GLM was handled by “glm” function in “lme4” package and  
359 model averaging by “model.avg” function in “MuMIn” package. Overdispersion checking  
360 was carried out by “dispersiontest” function in “AER” package.

361

## 362 **Results**

### 363 **Resources richness in the field, trace metals and diet richness**

364 Plant items occurred in the diet of 96.3% of wood mice, arthropod items in 65.9% and  
365 earthworm items in 29.3%. The occurrence of earthworm items was significantly lower in

366 forest landscape and significantly higher in urban landscape ( $\chi^2 = 7.619$ , p-value = 0.022;  
367 for details see SI5). The variations of the total richness in the diet and of the relative  
368 proportion of the three food item categories (plants, arthropods and earthworms) along the  
369 soil contamination gradient are shown in SI6 and SI7, respectively.

370 Number of plant MOTUs per mouse ranged from 0 to 7 and median was 2 MOTUs. It was  
371 significantly higher in autumn than in spring ( $W = 5394.5$ , p-value < 0.001). Number of  
372 arthropod MOTUs ranged from 0 to 9 and median was 1 MOTUs. In autumn, the number  
373 was significantly lower in age class I than in class II (Kruskal-Wallis  $\chi^2 = 7.972$ , p-value =  
374 0.047) (N.B. no mouse of class I was found in spring). Number of earthworm MOTUs ranged  
375 from 0 to 3 and median was 0 MOTUs. The number was significantly higher in urban area  
376 than in forest area in spring only (Kruskal-Wallis  $\chi^2 = 8.068$ , p-value = 0.018) (see SI8).

377 No single “best” model was evidenced and thus model averaging was applied for all sets of  
378 models. Complete results of modelling are shown in SI9. The final model for plant richness  
379 in the diet in buffer included mouse number, soil TM concentrations, richness in the field  
380 and the interaction between the last two for the three TM in the two seasons. Figure 1a and  
381 1b show plant richness in the diet according to plant richness in the field and Cd  
382 concentration in soils in spring, respectively. The pattern was similar for the other metals.

383 Influence of richness in the field on the diet richness was tiny in spring (Figure 1a), whereas  
384 diet richness decreased according to TM in soils regardless of the richness in the field (e.g.  
385 from 6 to 4 plant MOTUs along the range of TM concentration; Figure 1b).  $R^2_D$  was 0.301,  
386 0.313 and 0.307 for Cd, Pb and Zn, respectively. Such tendency changed in autumn (Figure  
387 1c and 1d for Cd, similar for other TM). Diet richness was negatively correlated to plant  
388 richness when soil contamination was low, while this correlation disappeared, or even  
389 reversed, when TM concentration increased (Figure 1c). A similar tendency was predicted  
390 for the relationship between diet richness and TM contamination in soils (Figure 1d): diet

391 richness was negatively correlated to TM concentration when plant richness was low and  
392 positively correlated when the plant richness was high.  $R^2_D$  was 0.624, 0.492 and 0.601 for  
393 Cd, Pb and Zn, respectively.

394 The model for invertebrate richness in the diet included all proposed explanatory variables  
395 in the two seasons, except the interaction between TMs in soils and richness of ground-  
396 dwelling invertebrates for the models for Cd and Zn in autumn. In spring, arthropod richness  
397 in the diet increased according to richness of both ground-dwelling and flying invertebrates  
398 when soil contamination was low. This correlation disappeared, then reversed, when TM  
399 concentration in soils increased. For example, our model for Cd predicted that arthropod  
400 MOTUs increased from 2 to 5 along the gradient of ground-dwelling invertebrate richness  
401 in the field when Cd concentration in soils was low, whereas arthropod MOTUs decreased  
402 from 5 to 3 when Cd concentration was high (Figure 2a; the pattern was similar for flying  
403 invertebrates). The relationship between arthropod richness in the diet and TM concentration  
404 in soils followed the same tendency (Figure 2b). In autumn, arthropod richness in the diet  
405 was explained almost only by mouse number. Neither TM concentration in soils nor richness  
406 of both types of invertebrates had sensible influence on diet richness (Figure 2c and 2d).  $R^2_D$   
407 for Cd, Pb and Zn was respectively 0.502, 0.472 and 0.495 in spring, and 0.495, 0.504 and  
408 0.497 in autumn.

409 The averaging model for earthworm richness in the diet included mouse number and TM  
410 concentrations in soils in both seasons. Earthworm richness in the diet slightly decreased  
411 along TM gradient (e.g. difference of about 0.5 MOTUs; Figure 3a and 3b), and mouse  
412 number on trap line had a larger influence in autumn than in spring.  $R^2_D$  for Cd, Pb and Zn  
413 was respectively 0.101, 0.056 and 0.075 in spring, and 0.169, 0.158, 0.192 in autumn.

414

415 **Plant food selection**

416 We found 48 Grps in the food of the 246 mice. Sapindaceae, Salicaceae, Cornaceae,  
417 Adoxaceae, Asteraceae, and Rosaceae were plant taxa frequently eaten in our area (see SI10  
418 for details). Selection for some plant families was observed at the scale of all study sites  
419 together and at the two seasons: preference for Fagaceae and avoidance of Urticaceae.  
420 Furthermore, selected plant resources changed between the two seasons, for example  
421 preference for Salicaceae, Sapindaceae and Rosaceae in spring and preference for Poaceae,  
422 Adoxaceae, Asteraceae and Cornaceae in autumn (see SI11 for details).  
423 Model averaging was used for the final model for Salicaceae preference in spring (see SI12  
424 for complete results). The model included all variables: mouse number, plant richness in the  
425 field, TM concentration in soils and the interaction. Preference for Salicaceae was negatively  
426 correlated to plant richness in the field when soil contamination was low (e.g. from 0.6 to  
427 0.4; Figure 4a). On the other hand, preference for Salicaceae disappeared along the gradient  
428 of TM concentration in soils, whatever plant richness in the field was (e.g. preference  
429 probability for Salicaceae decreased from 0.6 to 0; Figure 4b).  $R^2_D$  of the final model was  
430 0.280, 0.165 and 0.271 for Cd, Pb and Zn, respectively.

431

## 432 **Discussion**

433 We first hypothesised that soil TM contamination would reduce diet richness of wood mouse  
434 due to adverse effect of TM on resources' richness in the field. Our study revealed negative  
435 correlations between diet richness and TM in soils. This was true for both earthworms and  
436 plants in spring and in autumn (although only at low and medium levels of plant richness in  
437 autumn), and for arthropods in spring when richness in the field was high. On the contrary,  
438 positive correlations were observed, for plants in autumn when plant richness in the field  
439 was high, and for arthropods in spring when arthropod richness in the field was low. A  
440 negative relationship between diet richness and pollution by TM is expected if resource

441 richness in the field is negatively affected by TM concentrations. Literature about the effects  
442 of pollution on various taxonomic groups suggests a global negative effect of pollution on  
443 the richness of organisms, and an absence or a positive correlation for a few taxa groups or  
444 situations. Although decreasing richness of earthworms with increasing TM concentrations  
445 gradients in soils has been widely reported (Spurgeon & Hopkin, 1996, Spurgeon & Hopkin,  
446 1999; Pérès et al., 2011), several studies have reported that relationships between species  
447 richness of soil macro-fauna and TM contamination is not straightforward (Nahmani &  
448 Lavelle, 2002; Gillet & Ponge, 2003; Migliorini, et al., 2004). Moreover, several studies  
449 showed that community composition was affected by soil TM concentrations. Tolerance to,  
450 and/or accumulation capacity of metals differ according to taxa, organs, and metals (Das et  
451 al., 1997; Broadley et al., 2007; Pourrut et al., 2011). Lower resistance to toxic effects can  
452 make metal sensitive plants disappear in contaminated areas, which may favour metal  
453 tolerant ones, sometimes without modification of taxonomic richness. Such modification of  
454 community composition has been reported by several authors in plants (Banášová et al.,  
455 2006; Strandberg et al., 2006; Dazy et al., 2009) and in invertebrates (Nahmani & Lavelle,  
456 2002; Migliorini et al., 2004; Babin-Fenske & Anand, 2011). In our study area, we found no  
457 correlation or even positive trends between TM concentration in soils and both plant and  
458 invertebrate richness in the field (unpublished data). However, there were confounding  
459 factors in our site like the anthropogenic increase of habitat diversity (see Douay et al., 2009)  
460 and possibly of plant richness along the gradient of contamination on the Metaleurop Nord  
461 site. Community composition also differed according to soil contamination levels or other  
462 soil properties affecting metal bioavailability (unpublished data). Difference in resource  
463 composition could alter resource availability and thus modify feeding behaviour and diet  
464 composition. Few studies documented the effects of pollution on the diet of terrestrial  
465 vertebrates, most of them dealing with diet composition rather than richness or diversity.

466 Clare et al. (2014), describing the diet of insectivorous bats *Myotis lucifugus* in different  
467 locations, showed a trend of lower diet species richness in area of lower habitat quality  
468 (acidification or organic pollution). Eeva et al. (2005) showed a significant difference in diet  
469 composition of nestlings of two insectivorous birds, *Parus major* and *Ficedula hypoleuca*,  
470 between unpolluted and TM polluted sites, reflecting differences in resources composition,  
471 phenology and feeding ecology among the two bird species: in polluted areas where  
472 caterpillars were less abundant, both species ate more beetles and different larvae of flying  
473 insects but less moth. The study of Heroldová (2002) demonstrated that the diet of field vole  
474 *Microtus agrestis* in forest clearings caused by air pollution was dominated by grasses,  
475 reflecting higher abundance of grasses and a poorer diversity of food supply in clearings.  
476 Thus, it is suggested that diet richness (as demonstrated in the present study) and/or  
477 composition can be affected by pollution induced changes in trophic resources.

478 We also hypothesized that TM contamination would affect food selection by wood mouse.  
479 Our model predicts that preference probability for Salicaceae in spring in the highest  
480 contaminated sites decreases to become almost null. Avoidance of TM contaminated soil or  
481 food has been shown in invertebrates but has rarely been investigated in vertebrates, and is  
482 particularly hard to prove under field conditions in comparison with controlled experiments.  
483 This behaviour has been observed under laboratory conditions for some invertebrates like  
484 isopods or small moths (Odendaal & Reinecke, 1999; Zidar et al., 2004; Scheirs et al., 2006),  
485 and for the wood mouse (Beernaert et al., 2008), but mechanisms of such aversion are not  
486 yet elucidated. In Beernaert et al. (2008), wood mice chose acorns (*Quercus robur*,  
487 Fagaceae) sampled from unpolluted sites rather than ones from metal polluted sites (mainly  
488 by Cd, copper, Pb, and Zn), but such avoidance behaviour was no longer detected under field  
489 conditions, perhaps due to ecological co-factors such as food shortage, and/or the presence  
490 of competitors and predators. Avoidance of contaminated food, leading to shifts in diet, can

491 be one of the mechanisms allowing animals to mitigate their exposure in highly polluted  
492 environments (Gillet & Ponge, 2003). Actually, Salicaceae family is considered as TM  
493 accumulator and resistant to highly TM contaminated soils (Pulford & Watson, 2003).  
494 Migeon et al. (2009) showed that Salicaceae family in the surroundings of Metaleurop Nord,  
495 composed of two genera *Salix* and *Populus*, accumulated larger amount of Cd and Zn in their  
496 shoot parts than the other families like Fagaceae or Sapindaceae. These two last are also  
497 preferred plant families in our study and known as common food of wood mice (i.e. Watts,  
498 1968). However, mechanisms of changes in preference remains unclear. The nutritional  
499 quality of metal contaminated plant organs has been shown to be lowered, with for instance  
500 a decrease of sugar or protein concentrations (Scheirs et al., 2006; Beernaert et al., 2008),  
501 features which play an important role for food choice in rodents (Butet, 1990; Kerley &  
502 Erasmus, 1991). It is also possible that animals might identify metal contaminated plants due  
503 to metal-induced metabolic compounds (Poschenrieder et al., 2006), largely known as  
504 compounds limiting feeding (Foley & Moore, 2005). Dearing et al (2000) reported that plant  
505 secondary compounds might play more important role than nutrient level in the diet of the  
506 generalist woodrat, *Neotoma albigula*. It is therefore supposed that mice could directly or  
507 indirectly detect noxious substances (plant toxins or TM) and avoid ingesting such items.  
508 Also, Behmer et al. (2005) demonstrated from their experiments using nutritionally balanced  
509 synthetic food that high Zn concentration *per se* altered the feeding behaviour of the desert  
510 locust *Schistocerca gregaria*. Whatever the mechanism, the hyperaccumulation of metals in  
511 plant tissues contribute to defensive effects of plants against natural enemies, including  
512 invertebrate herbivory (Poschenrieder et al., 2006; Boyd, 2007). Such defensive effect has  
513 not been reported against vertebrates so far, but the disappearance of the preference for  
514 Salicaceae family along TM contamination in soils could be associated to its high metal  
515 accumulation capacity. Furthermore, plant richness in the diet in autumn increased along the

516 soil TM concentration gradient when richness in the field was high. Apart from changes of  
517 resource composition, this pattern could be explained, at least partly, by the nutrient balance  
518 hypothesis or the detoxification limitation hypothesis: if nutritional quality of all plant  
519 resources was altered (i.e. if nutritional quality was not clearly contrasted among resources)  
520 diet richness of animals could increase for completing nutrient and/or for avoiding loads of  
521 similar secondary compounds.

522 Contrary to plants, selection for invertebrate food could not be analysed in part because of  
523 the primer used for arthropods and molluscs: its short size is more reliable for the analysis  
524 of degraded DNA in the case of diet (Taberlet et al., 2012) but the precision in taxonomic  
525 assignment was low. We also did not sample earthworms in the field. Moreover, it is difficult  
526 to determine selection for invertebrate resources because of their mobility. Foraging strategy  
527 of predators can be modified according to mobility of preys (Sih & Christensen, 2001),  
528 suggesting that relative quantity in the field is not the only factor to consider when analysing  
529 mobile prey selection. Even though we supposed that decreased earthworm diet richness  
530 might be linked to their richness in the field, invertebrate food selection in wood mice  
531 remains as a further challenge.

532 An important feature of our results is the strong seasonal difference of patterns, for both food  
533 selection and correlations with TM contamination. In temperate ecosystems, seasons have a  
534 key influence on the dynamics of ecosystems and on the various aspects of biology and  
535 ecology of living organisms. Seasonal difference of patterns might result from the adaptation  
536 of wood mice feeding behaviour to phenology of resources. Indeed, wood mice consume  
537 mainly seeds and other reproductive organs (Watts, 1968; Butet, 1986a), seeking high energy  
538 or protein-rich tissues (Butet, 1990). Although parts of plant resources actually consumed  
539 could not be distinguished by using DNA metabarcoding, we supposed that parts of plants  
540 preferably consumed have changed between spring and autumn. Moreover, the nutritional

541 composition and quality of plants and invertebrates are likely to vary according to seasons.  
542 Also, resources functionally important for wood mice might have a strong phenology, such  
543 as mushrooms in autumn. However, the molecular primers we used did not target mushrooms,  
544 which prevents further conclusion. Finally, TM accumulation in organisms is also subject to  
545 seasonality at contaminated site (i.e. Hunter et al, 1987b; Migeon et al., 2009). Seasonal  
546 variation of TM accumulation pattern can be related to growth season or physiological  
547 changes of each taxa (Kabata-Pendias, 2011; Hussein et al., 2006). Disentangling the  
548 combined effects of these factors on the patterns of interest in this study is beyond our present  
549 scope, but it is clear that seasonality is a key feature of the interplay between pollution,  
550 resource dynamics and consumer feeding behaviour.

551 Our study sheds light on the effects of soil TM contamination on patterns of feeding  
552 behaviour of a generalist consumer, by applying DNA metabarcoding for diet identification.  
553 We underline potential altering effects of TM on the diet richness, which possibly partly  
554 results from modifying food selection, at least for plant resources. Such modification of diet  
555 related to habitat contamination might lead to a different pattern of contaminant exposure in  
556 the animals. We hence suggest that ecosystem pollution should now be considered as a driver  
557 of changes in food web structure and dynamics in time and space.

558

### 559 **Acknowledgements**

560 This study was financially supported by the project BIOTROPH, co-funded by the *Agence*  
561 *De l'Environnement et de la Maîtrise de l'Energie* (ADEME; contract No.1172C0030) and  
562 the *Conseil Régional du Nord-Pas de Calais* (CRNPC; orders No.12000921 and 14001044;  
563 joint call with the *Fondation pour la Recherche sur la Biodiversité*). The first author was  
564 also financially supported by a grant from the *Conseil Régional de Franche-Comté* (contract  
565 No. 2015C-06107). The authors gratefully thank Cécile Grand from ADEME for fruitful

566 scientific discussions. We also thank Eva Bellemain and Alice Valentini from SPYGEN  
567 Company for their help in DNA analyses and interpretation. We finally thank François Gillet,  
568 Nadia Crini, Dominique Rieffel and Anne-Sophie Prudent for their precious assistance.

569 **References:**

- 570
- 571 Abt, K. F., & Bock, W. F. (1998). Seasonal variations of diet composition in farmland field  
572 mice *Apodemus* spp. and bank voles *Clethrionomys glareolus*. *Acta Theriologica*, *43*,  
573 379–389.
- 574 Babin-Fenske, J., & Anand, M. (2011). Patterns of insect communities along a stress gradient  
575 following decommissioning of a Cu–Ni smelter. *Environmental Pollution*, *159*,  
576 3036–3043.
- 577 Bailey, B. J. R. (1980). Large sample simultaneous confidence intervals for the multinomial  
578 probabilities based on transformation of the cell frequencies. *Technometrics*, *22*,  
579 583–589.
- 580 Banášová, V., Horak, O., Čiamporová, M., Nadubinská, M., & Lichtscheidl, I. (2006). The  
581 vegetation of metalliferous and non-metalliferous grasslands in two former mine  
582 regions in Central Slovakia. *Biologia*, *61*, 433–439.
- 583 Beernaert, J., Scheirs, J., Van Den Brande, G., Leirs, H., Blust, R., De Meulenaer, B., ...  
584 Verhagen, R. (2008). Do wood mice (*Apodemus sylvaticus* L.) use food selection as  
585 a means to reduce heavy metal intake? *Environmental Pollution*, *151*, 599–607.
- 586 Behmer, S. T., Lloyd, C. M., Raubenheimer, D., Stewart-Clark, J., Knight, J., Leighton, R.  
587 S., ... Smith, J. A. C. (2005). Metal hyperaccumulation in plants: mechanisms of  
588 defence against insect herbivores. *Functional Ecology*, *19*, 55–66.
- 589 Bienert, F., De Danieli, S., Miquel, C., Coissac, E., Poillot, C., Brun, J.-J., & Taberlet, P.  
590 (2012). Tracking earthworm communities from soil DNA. *Molecular Ecology*, *21*,  
591 2017–2030.
- 592 Bohmann, K., Monadjem, A., Lehmkuhl Noer, C., Rasmussen, M., Zeale, M. R. K., Clare,  
593 E., ... Gilbert, M. T. P. (2011). Molecular Diet Analysis of Two African Free-Tailed  
594 Bats (Molossidae) Using High Throughput Sequencing. *PLoS ONE*, *6*, e21441.

- 595 Boyd, R. S. (2007). The defense hypothesis of elemental hyperaccumulation: status,  
596 challenges and new directions. *Plant and Soil*, 293, 153–176.
- 597 Braun-Blanquet, J., Roussine, N., & Nègre, R. (1952). *Les groupements végétaux de la*  
598 *France méditerranéenne*. CNRS: Direction de la Carte des Groupements Végétaux  
599 de l'Afrique du Nord.
- 600 Broadley, M. R., White, P. J., Hammond, J. P., Zelko, I., & Lux, A. (2007). Zinc in plants.  
601 *New Phytologist*, 173, 677–702.
- 602 Burnham, K. P., & Anderson, D. R. (2002). *Model selection and multimodel inference: a*  
603 *practical information-theoretic approach* (2<sup>nd</sup> ed). New York: Springer.
- 604 Butet, A. (1986a). Régime alimentaire d'une population de mulots sylvestres (*Apodemus*  
605 *sylvaticus* L., 1758), dans une lande xéro-mésophile en cours de recolonisation  
606 végétale. *Bulletin D'Ecologie*, 17, 21–37.
- 607 Butet, A. (1986b). Stratégie d'utilisation et déterminisme des choix alimentaires d'un  
608 rongeur polyphage (*Apodemus sylvaticus*, L.): l'approche énergétique. *Acta*  
609 *Oecologica*, 7, 243–262.
- 610 Butet, A. (1990). Teneur azotée des ressources et choix trophiques du Mulot sylvestre  
611 (*Apodemus sylvaticus*) dans un système oligotrophe. *Canadian Journal of Zoology*,  
612 68, 26–31.
- 613 Cameron, A. C., & Trivedi, P. K. (1990). Regression-based Tests for Overdispersion in the  
614 Poisson Model. *Journal of Econometrics*, 46, 347–364.
- 615 Cherry, S. (1996). A comparison of confidence interval methods for habitat use-availability  
616 studies. *The Journal of Wildlife Management*, 60, 653–658.
- 617 Clare, E. L., Symondson, W. O. C., Broders, H., Fabianek, F., Fraser, E. E., MacKenzie, A.,  
618 ... Reimer, J. P. (2014). The diet of *Myotis lucifugus* across Canada: assessing  
619 foraging quality and diet variability. *Molecular Ecology*, 23, 3618–3632.

- 620 Coulon, J. (2003). Les Bembidiina de la faune de France. Clés d'identification commentées  
621 (Coleoptera Carabidae Trechinae) (Première partie). *Bulletin mensuel de la Société*  
622 *linnéenne de Lyon*, 72, 256–272.
- 623 Crampton, L. H., Longland, W. S., Murphy, D. D., & Sedinger, J. S. (2011). Food abundance  
624 determines distribution and density of a frugivorous bird across seasons. *Oikos*, 120,  
625 65–76.
- 626 Das, P., Samantaray, S., & Rout, G. R. (1997). Studies on cadmium toxicity in plants: a  
627 review. *Environmental Pollution*, 98, 29–36.
- 628 Dazy, M., Béraud, E., Cotelle, S., Grévilliot, F., Féraud, J.-F., & Masfaraud, J.-F. (2009).  
629 Changes in plant communities along soil pollution gradients: Responses of leaf  
630 antioxidant enzyme activities and phytochelatin contents. *Chemosphere*, 77, 376–  
631 383.
- 632 De Barba, M., Miquel, C., Boyer, F., Mercier, C., Rioux, D., Coissac, E., & Taberlet, P.  
633 (2013). DNA metabarcoding multiplexing and validation of data accuracy for diet  
634 assessment: application to omnivorous diet. *Molecular Ecology Resources*, 14, 306–  
635 323.
- 636 Deagle, B. E., Thomas, A. C., Shaffer, A. K., Trites, A. W., & Jarman, S. N. (2013).  
637 Quantifying sequence proportions in a DNA-based diet study using Ion Torrent  
638 amplicon sequencing: which counts count? *Molecular Ecology Resources*, 13, 620–  
639 633.
- 640 Dearing, M. D., Mangione, A. M., & Karasov, W. H. (2000). Diet breadth of mammalian  
641 herbivores: nutrient versus detoxification constraints. *Oecologia*, 123, 397–405.
- 642 Douay, F., Pruvot, C., Waterlot, C., Fritsch, C., Fourier, H., Loriette, A., ... Scheifler, R.  
643 (2009). Contamination of woody habitat soils around a former lead smelter in the  
644 North of France. *Science of The Total Environment*, 407, 5564–5577.

- 645 Douay, F., Pruvot, C., Roussel, H., Ciesielski, H., Fourrier, H., Proix, N., & Waterlot, C.  
646 (2008). Contamination of Urban Soils in an Area of Northern France Polluted by  
647 Dust Emissions of Two Smelters. *Water, Air, and Soil Pollution*, 188, 247–260.
- 648 Dudman, A. A., & Richards, A. J. (1997). *Dandelions of Great Britain and Ireland*,  
649 *Handbook 9*. London: Botanical Society of the British Isles.
- 650 Eeva, T., Ryömä, M., & Riihimäki, J. (2005). Pollution-related changes in diets of two  
651 insectivorous passerines. *Oecologia*, 145, 629–639.
- 652 Eisler, R. (2000). *Handbook of chemical risk assessment: health hazards to humans, plants*,  
653 *and animals. Volume 1, metals*. Boca Raton, Florida: Lewis.
- 654 EMEP (2013). *Long term Changes of Heavy Metal Transboundary Pollution of the*  
655 *Environment (1990-2010)* (No. 2/2013).
- 656 Foley, W. J., & Moore, B. D. (2005). Plant secondary metabolites and vertebrate herbivores  
657 – from physiological regulation to ecosystem function. *Current Opinion in Plant*  
658 *Biology*, 8, 430–435.
- 659 Forel, J., & Leplat, J. (2001). *Faune des carabiques de France* (Vol. 1). Magellanes.
- 660 Freeland, W. J., & Janzen, D. H. (1974). Strategies in herbivory by mammals: the role of  
661 plant secondary compounds. *The American Naturalist*, 108, 269–289.
- 662 Fritsch, C., Cœurassier, M., Giraudoux, P., Raoul, F., Douay, F., Rieffel, D., ... Scheifler,  
663 R. (2011). Spatially Explicit Analysis of Metal Transfer to Biota: Influence of Soil  
664 Contamination and Landscape. *PLoS ONE*, 6, e20682.
- 665 Fritsch, C., Giraudoux, P., Cœurassier, M., Douay, F., Raoul, F., Pruvot, C., ... Scheifler, R.  
666 (2010). Spatial distribution of metals in smelter-impacted soils of woody habitats:  
667 Influence of landscape and soil properties, and risk for wildlife. *Chemosphere*, 81,  
668 141–155.
- 669 Gillet, S., & Ponge, J.-F. (2003). Changes in species assemblages and diets of Collembola

670 along a gradient of metal pollution. *Applied Soil Ecology*, 22, 127–138.

671 Hansson, L. (1985). The food of bank voles, wood mice and yellow-necked mice.  
672 *Symposium of the Zoological Society of London*, 55, 141–168.

673 Harwood, J. D., Phillips, S. W., Sunderland, K. D., & Symondson, W. O. C. (2001).  
674 Secondary predation: quantification of food chain errors in an aphid–spider–carabid  
675 system using monoclonal antibodies. *Molecular Ecology*, 10, 2049–2057.

676 Heroldová, M. (2002). Food selection of *Microtus agrestis* in air-pollution affected clearings  
677 in the Beskydy Mts, Czech Republic. *Folia Zoologica*, 51, 83–91.

678 Hunter, B. A., & Johnson, M. S. (1982). Food chain relationships of copper and cadmium in  
679 contaminated grassland ecosystems. *Oikos*, 38, 108–117.

680 Hunter, B. A., Johnson, M. S., & Thompson, D. J. (1987a). Ecotoxicology of copper and  
681 cadmium in a contaminated grassland ecosystem III. Small mammals. *Journal of*  
682 *Applied Ecology*, 24, 601–614.

683 Hunter, B. A., Johnson, M. S., & Thompson, D. J. (1987b). Ecotoxicology of copper and  
684 cadmium in a contaminated grassland ecosystem II. Invertebrates. *Journal of Applied*  
685 *Ecology*, 24, 587–599.

686 Hussein, M. A., Obuid-Allah, A. H., Mohammad, A. H., Scott-Fordsmand, J. J., & Abd El-  
687 Wakeil, K. F. (2006). Seasonal variation in heavy metal accumulation in subtropical  
688 population of the terrestrial isopod, *Porcellio laevis*. *Ecotoxicology and*  
689 *Environmental Safety*, 63, 168–174.

690 Jeannel, R. (1941). *Coléoptères carabiques*. Paris: Librairie de la faculté des sciences.

691 Johnson, D. H. (1980). The Comparison of Usage and Availability Measurements for  
692 Evaluating Resource Preference. *Ecology*, 61, 65–71.

693 Kabata-Pendias, A. (2011). *Trace elements in soils and plants* (4<sup>th</sup> ed). Boca Raton: CRC  
694 Press.

- 695 Kerley, G. I. H., & Erasmus, T. (1991). What do mice select for in seeds? *Oecologia*, *86*,  
696 261–267.
- 697 Klare, U., Kamler, J. F., & Macdonald, D. W. (2011). A comparison and critique of different  
698 scat-analysis methods for determining carnivore diet: Comparison of scat-analysis  
699 methods. *Mammal Review*, *41*, 294–312.
- 700 Kozlov, M. V., & Zvereva, E. L. (2011). A second life for old data: Global patterns in  
701 pollution ecology revealed from published observational studies. *Environmental*  
702 *Pollution*, *159*, 1067–1075.
- 703 Lambinon, J., Delvosalle, L., & Duvigneaud, J. (2004). *Nouvelle flore de la Belgique, du*  
704 *Grand-Duché de Luxembourg, du Nord de la France et des Régions voisines* (5th ed).  
705 Meise: Jardin botanique national de Belgique.
- 706 Lewis, C. E., Clark, T. W., & Derting, T. L. (2001). Food selection by the white-footed mouse  
707 (*Peromyscus leucopus*) on the basis of energy and protein contents. *Canadian*  
708 *Journal of Zoology*, *79*, 562–568.
- 709 Lukacs, P. M., Burnham, K. P., & Anderson, D. R. (2010). Model selection bias and  
710 Freedman’s paradox. *Annals of the Institute of Statistical Mathematics*, *62*, 117–125.
- 711 MacArthur, R. H., & Pianka, E. R. (1966). On optimal use of a patchy environment. *The*  
712 *American Naturalist*, *100*, 603–609.
- 713 Manly, B. F. L., McDonald, L., Thomas, D., McDonald, T. L., & Erickson, W. P. (2002).  
714 *Resource selection by animals: statistical design and analysis for field studies* (2<sup>nd</sup>  
715 ed). Dordrecht: Kluwer Academic.
- 716 Migeon, A., Richaud, P., Guinet, F., Chalot, M., & Blaudez, D. (2009). Metal Accumulation  
717 by Woody Species on Contaminated Sites in the North of France. *Water, Air, and Soil*  
718 *Pollution*, *204*, 89–101.
- 719 Migliorini, M., Pigino, G., Bianchi, N., Bernini, F., & Leonzio, C. (2004). The effects of

720 heavy metal contamination on the soil arthropod community of a shooting range.  
721 *Environmental Pollution*, 129,

722 Montgomery, S. S. J., & Montgomery, W. I. (1990). Intrapopulation variation in the diet of  
723 the wood mouse *Apodemus sylvaticus*. *Journal of Zoology*, 222, 641–651.

724 Nahmani, J., & Lavelle, P. (2002). Effects of heavy metal pollution on soil macrofauna in a  
725 grassland of Northern France. *European Journal of Soil Biology*, 38, 297–300.

726 Nahmani, J., Lavelle, P., Lapied, E., & van Oort, F. (2003). Effects of heavy metal soil  
727 pollution on earthworm communities in the north of France: The 7<sup>th</sup> international  
728 symposium on earthworm ecology. Cardiff. Wales. 2002. *Pedobiologia*, 47, 663–  
729 669.

730 Odendaal, J. P., & Reinecke, A. J. (1999). Short-Term Toxicological Effects of Cadmium on  
731 the Woodlouse, *Porcellio laevis* (Crustacea, Isopoda). *Ecotoxicology and*  
732 *Environmental Safety*, 43, 30–34.

733 Pérès, G., Vandebulcke, F., Guernion, M., Hedde, M., Beguiristain, T., Douay, F., ...  
734 Cluzeau, D. (2011). Earthworm indicators as tools for soil monitoring,  
735 characterization and risk assessment. An example from the national Bioindicator  
736 programme (France). *Pedobiologia*, 54S, S77–S87.

737 Polis, G. A., Anderson, W. B., & Holt, R. D. (1997). Toward an integration of landscape and  
738 food web ecology: the dynamics of spatially subsidized food webs. *Annual Review*  
739 *of Ecology and Systematics*, 28, 289–316.

740 Pompanon, F., Deagle, B. E., Symondson, W. O. C., Brown, D. S., Jarman, S. N., & Taberlet,  
741 P. (2012). Who is eating what: diet assessment using next generation sequencing.  
742 *Molecular Ecology*, 21, 1931–1950.

743 Poschenrieder, C., Tolrà, R., & Barceló, J. (2006). Can metals defend plants against biotic  
744 stress? *Trends in Plant Science*, 11, 288–295.

- 745 Pourrut, B., Shahid, M., Dumat, C., Winterton, P., & Pinelli, E. (2011). Lead Uptake, Toxicity,  
746 and Detoxification in Plants. *Reviews of Environmental Contamination and*  
747 *Toxicology*, 213, 113–136.
- 748 Pulford, I. D., & Watson, C. (2003). Phytoremediation of heavy metal-contaminated land by  
749 trees—a review. *Environment International*, 29, 529–540.
- 750 Pyke, G. H., Pulliam, H. R., & Charnov, E. L. (1977). Optimal foraging: a selective review  
751 of theory and tests. *Quarterly Review of Biology*, 52, 137–154.
- 752 Quéré, J.-P., & Le Louarn, H. (2011). *Les rongeurs de France : Faunistique et biologie* (2<sup>nd</sup>  
753 ed). Versailles: Quae.
- 754 Rogival, D., Scheirs, J., & Blust, R. (2007). Transfer and accumulation of metals in a soil–  
755 diet–wood mouse food chain along a metal pollution gradient. *Environmental*  
756 *Pollution*, 145, 516–528.
- 757 Scheirs, J., Vandevyvere, I., Wollaert, K., Blust, R., & De Bruyn, L. (2006). Plant-mediated  
758 effects of heavy metal pollution on host choice of a grass miner. *Environmental*  
759 *Pollution*, 143, 138–145.
- 760 Schloss, P. D., Westcott, S. L., Ryabin, T., Hall, J. R., Hartmann, M., Hollister, E. B., ...  
761 Weber, C. F. (2009). Introducing mothur: Open-Source, Platform-Independent,  
762 Community-Supported Software for Describing and Comparing Microbial  
763 Communities. *Applied and Environmental Microbiology*, 75, 7537–7541.
- 764 Schmidt, K. A. (2000). Interactions between food chemistry and predation risk in fox  
765 squirrels. *Ecology*, 81, 2077–2085.
- 766 Schoener, T. W. (1971). Theory of feeding strategies. *Annual Review of Ecology and*  
767 *Systematics*, 2, 369–404.
- 768 Shore, R. F., & Rattner, B. A. (2001). *Ecotoxicology of Wild Mammals*. New York: John  
769 Wiley and Sons.

- 770 Sih, A., & Christensen, B. (2001). Optimal diet theory: when does it work, and when and  
771 why does it fail? *Animal Behaviour*, *61*, 379–390.
- 772 Smith, P. N., Cobb, G. P., Godard-Codding, C., Hoff, D., McMurry, S. T., Rainwater, T. R.,  
773 & Reynolds, K. D. (2007). Contaminant exposure in terrestrial vertebrates.  
774 *Environmental Pollution*, *150*, 41–64.
- 775 Soininen, E. M., Valentini, A., Coissac, E., Miquel, C., Gielly, L., Brochmann, C., ...  
776 Taberlet, P. (2009). Analysing diet of small herbivores: the efficiency of DNA  
777 barcoding coupled with high-throughput pyrosequencing for deciphering the  
778 composition of complex plant mixtures. *Frontiers in Zoology*, *6*, 16.
- 779 Spurgeon, D. J., & Hopkin, S. P. (1996). The effects of metal contamination on earthworm  
780 populations. *Applied Soil Ecology*, *4*, 147–160.
- 781 Spurgeon, D. J., & Hopkin, S. P. (1999). Seasonal variation in the abundance, biomass and  
782 biodiversity of earthworms in soils contaminated with metal emissions from a  
783 primary smelting works. *Journal of Applied Ecology*, *36*, 173–183.
- 784 Stephens, D. W., & Krebs, J. R. (1986). *Foraging Theory* (Princeton University Press).  
785 Princeton, New Jersey.
- 786 Sterckeman, T., Douay, F., Fourrier, H., & Proix, N. (2007). Référentiel pédo-géochimique  
787 du Nord-Pas de Calais. *Conseil Régional Du Nord-Pas de Calais*.
- 788 Sterckeman, T., Douay, F., Proix, N., Fourrier, H., & Perdrix, E. (2002). Assessment of the  
789 contamination of cultivated soils by eighteen trace elements around smelters in the  
790 North of France. *Water, Air, and Soil Pollution*, *135*, 173–194.
- 791 Strandberg, B., Axelsen, J. A., Pedersen, M. B., Jensen, J., & Attrill, M. J. (2006). Effect of  
792 a copper gradient on plant community structure. *Environmental Toxicology and*  
793 *Chemistry*, *25*, 743–753.
- 794 Sun, Y., Cai, Y., Huse, S. M., Knight, R., Farmerie, W. G., Wang, X., & Mai, V. (2012). A

795 large-scale benchmark study of existing algorithms for taxonomy-independent  
796 microbial community analysis. *Briefings in Bioinformatics*, *13*, 107–121.

797 Taberlet, P., Coissac, E., Pompanon, F., Gielly, L., Miquel, C., Valentini, A., ... Willerslev,  
798 E. (2007). Power and limitations of the chloroplast *trnL* (UAA) intron for plant DNA  
799 barcoding. *Nucleic Acids Research*, *35*(3), e14.

800 Taberlet, P., Coissac, E., Hajibabaei, M., & Rieseberg, L. H. (2012). Environmental DNA.  
801 *Molecular Ecology*, *21*, 1789–1793.

802 Tête, N., Afonso, E., Crini, N., Drouhot, S., Prudent, A.-S., & Scheifler, R. (2014). Hair as a  
803 noninvasive tool for risk assessment: Do the concentrations of cadmium and lead in  
804 the hair of wood mice (*Apodemus sylvaticus*) reflect internal concentrations?  
805 *Ecotoxicology and Environmental Safety*, *108*, 233–241.

806 Torres, K. C., & Johnson, M. L. (2001). Bioaccumulation of metals in plants, arthropods,  
807 and mice at a seasonal wetland. *Environmental Toxicology and Chemistry*, *20*, 2617–  
808 2626.

809 Trautner, J., & Geigenmueller, K. (1987). *Tiger Beetles, Ground Beetles (Illustrated Key to*  
810 *the Cicindelidae and Carabidae of Europe)*. Aichtal: Joseph Margraf.

811 Valentini, A., Pompanon, F., & Taberlet, P. (2009a). DNA barcoding for ecologists. *Trends*  
812 *in Ecology & Evolution*, *24*, 110–117.

813 Valentini, A., Miquel, C., Nawaz, M. A., Bellemain, E., Coissac, E., Pompanon, F., ...  
814 Taberlet, P. (2009b). New perspectives in diet analysis based on DNA barcoding and  
815 parallel pyrosequencing: the *trnL* approach. *Molecular Ecology Resources*, *9*, 51–60.

816 van den Brink, N., Lammertsma, D., Dimmers, W., Boerwinkel, M.-C., & van der Hout, A.  
817 (2010). Effects of soil properties on food web accumulation of heavy metals to the  
818 wood mouse (*Apodemus sylvaticus*). *Environmental Pollution*, *158*, 245–251.

819 Vanderpe, E., & Verhagen, R. (1980). An age reference model for the Wood mouse

- 820 *Apodemus sylvaticus* (Linnaeus, 1758), by use of the lens technique. *Annales de La*  
821 *Société Royale Zoologique de Belgique*, 109, 133–140.
- 822 Visser, M. E., & Both, C. (2005). Shifts in phenology due to global climate change: the need  
823 for a yardstick. *Proceedings of the Royal Society B: Biological Sciences*, 272, 2561–  
824 2569.
- 825 Walker, C. H., Hopkin, S. P., Sibly, R. M., & Peakall, D. B. (2012). *Principles of*  
826 *Ecotoxicology* (4<sup>th</sup> ed). Boca Raton: CRC Press.
- 827 Waterlot, C., Pruvot, C., Bidar, G., Fritsch, C., De Vaufleury, A., Scheifler, R., & Douay, F.  
828 (2016). Prediction of Extractable Cd, Pb and Zn in Contaminated Woody Habitat  
829 Soils Using a Change Point Detection Method. *Pedosphere*, 26, 282–298.
- 830 Watts, C. H. S. (1968). The Foods Eaten by Wood Mice (*Apodemus sylvaticus*) and Bank  
831 Voles (*Clethrionomys glareolus*) in Wytham Woods, Berkshire. *Journal of Animal*  
832 *Ecology*, 37, 25–41.
- 833 Westoby, M. (1978). What are the biological bases of varied diets? *The American Naturalist*,  
834 112, 627–631.
- 835 White, T. C. R. (2008). The role of food, weather and climate in limiting the abundance of  
836 animals. *Biological Reviews*, 83, 227–248.
- 837 Zidar, P., Drobne, D., Štrus, J., Van Gestel, C. A. M., & Donker, M. (2004). Food selection  
838 as a means of Cu intake reduction in the terrestrial isopod *Porcellio scaber* (Crustacea,  
839 Isopoda). *Applied Soil Ecology*, 25, 257–265.
- 840 Zuur, A. F. (2009). *Mixed effects models and extensions in ecology with R*. New York:  
841 Springer.

842

843 **Data Accessibility:**

844 DNA sequence data, soil TM concentration in buffers, resources availability in buffers, and  
845 wood mouse data are available from the metadata portal “dat@osu”;  
846 doi:10.25666/DATAOSU-2018-07-17.

847

848 **Supporting Information:**

849 SI1: Information for each study site about TM contamination and dominant landscape  
850 feature, as well as richness of plants and invertebrates observed in the field.

851 SI2: Inventory of plant and invertebrate taxa observed in each site.

852 SI3: Sequence data of plant, arthropod and earthworm MOTUs for each mouse.

853 SI4: Sequence data of sequence groups for each mouse.

854 SI5: Frequency of occurrence of plant, arthropod and earthworm items in wood mouse food  
855 across several environmental and biological factors.

856 SI6: Variation of the total richness in the diet along the soil contamination gradient.

857 SI7: Relative proportion of the three food item categories (plants, arthropods and  
858 earthworms) along the soil contamination gradient.

859 SI8: Plant, arthropod and earthworm richness in the diet of wood mouse across several  
860 environmental and biological factors.

861 SI9: Complete data on models of richness in the diet.

862 SI10: Complete data on number and frequency of occurrence of plant food item (sequence  
863 group).

864 SI11: Information on food selection for plant families corresponding to sequence groups in  
865 the wood mouse in spring and in autumn.

866 SI12: Complete results on plant food selection modelling at buffer scale.

867

868 **Author contributions:**

869 F.R. and R.S. conceived the study; S.O., R.S., C.F. and F.R. designed the study; R.S., C.F.,  
870 F.M., T.C. and F.R. performed field samplings and laboratory treatments; B.V. performed  
871 bioinformatics treatments; S.O. performed the statistical analyses; S.O., F.R., R.S. and C.F.  
872 interpreted the results. S.O. wrote the manuscript under the supervision of F.R, R.S. and C.F.

873

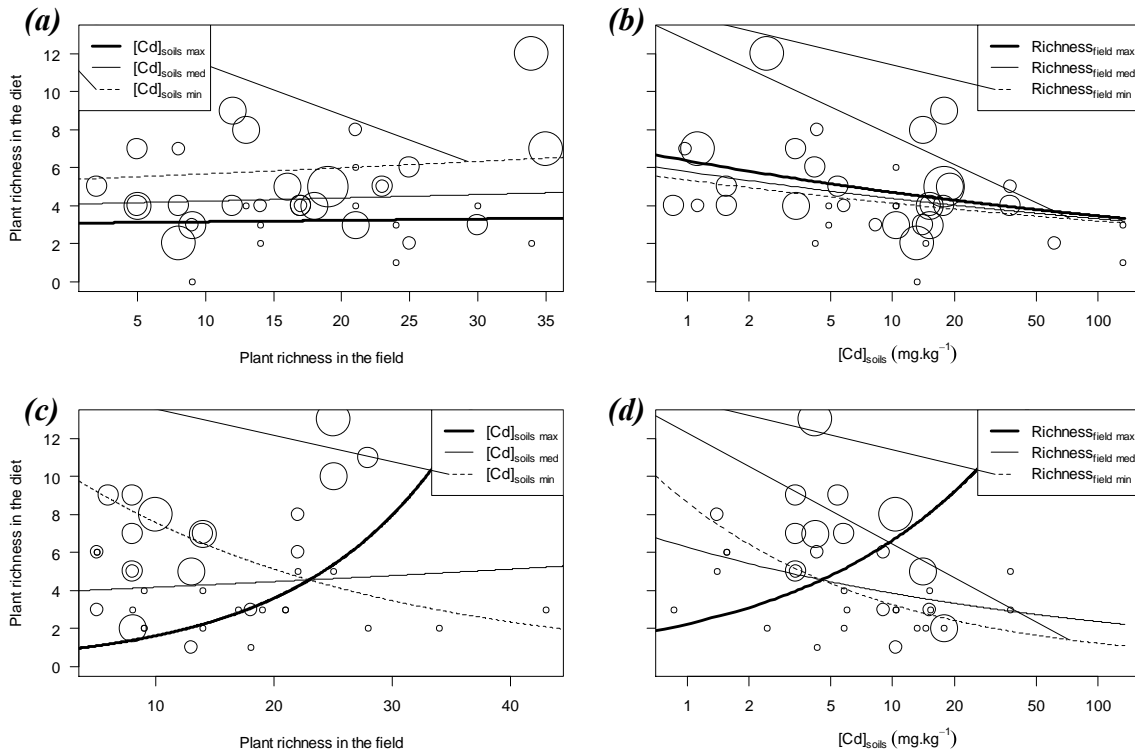
874  
875

*Table 1: Characteristics of the three primers used to amplify wood mouse stomach content DNA samples.*

Taxonomic group	DNA type	DNA region	Primer name	Primer sequence 5'-3'	Reference
Plants	Chloroplast	trnL (UAA)	g ( <i>forward</i> ) h ( <i>reverse</i> )	GGGCAATCCTGAGCCAA CCATTGAGTCTCTGCACCTATC	Taberlet <i>et al.</i> , 2007
Arthropods & molluscs	Mitochondrial	16S mtDNA	16SMAV-F 16SMAV-R	CCAACATCGAGGTCRYAA ARTTACYNTAGGGATAACAG	De Barba <i>et al.</i> , 2013
Earthworms	Mitochondrial	16S mtDNA	ewD ( <i>forward</i> ) ewE ( <i>reverse</i> )	ATTCGGTTGGGGCGACC CTGTTATCCCTAAGGTAGCTT	Bienert <i>et al.</i> , 2012

876

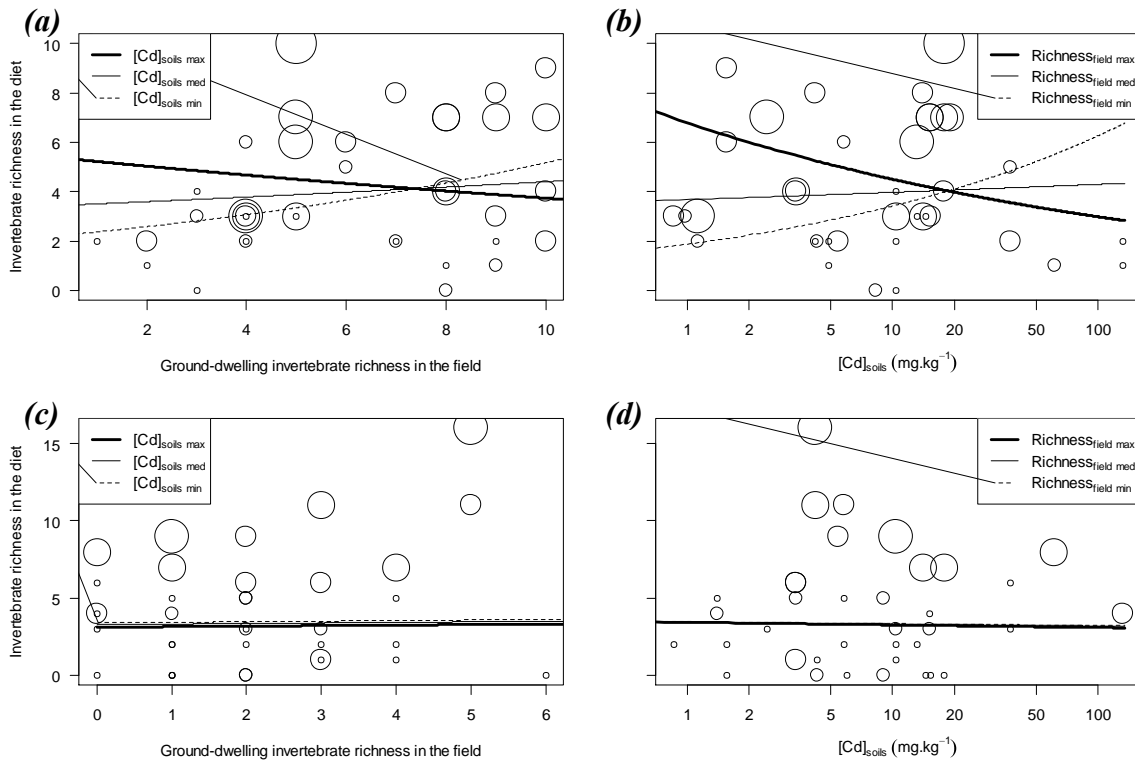
877 *Figure 1: Plant richness in the diet of the wood mouse along the gradient of plant richness*  
 878 *in the field and Cd concentration in soils, in spring (a and b) and in autumn (c and d). Each*  
 879 *point represents a buffer and point size is proportional to the number of mice captured in a*  
 880 *given buffer. Values predicted from models are indicated by three lines: dotted, normal and*  
 881 *bold lines correspond to minimal, median and maximum value of Cd concentration in soils*  
 882 *(a and c) or plants richness in the field (b and d), respectively. The number of mice is fixed*  
 883 *to the median value for drawing the lines (three mice in spring and two mice in autumn).*  
 884 *The patterns are similar for Pb and Zn concentrations in soils.*



885

886  
887

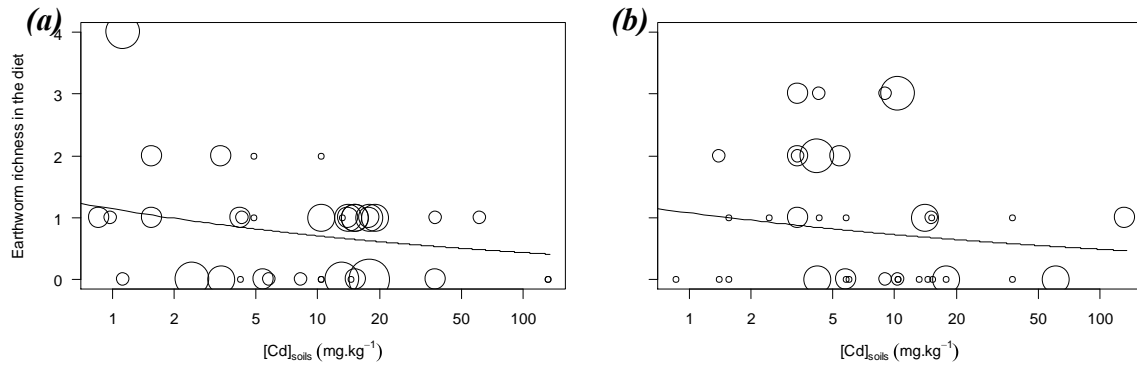
888 *Figure 2: Invertebrate richness in the diet of the wood mouse along the gradient of ground-*  
 889 *dwelling invertebrate richness in the field and Cd concentration in soils in spring (a and b)*  
 890 *and in autumn (c and d). Each point represents a buffer and point size is proportional to the*  
 891 *number of mice captured in a given buffer. Values predicted from models are indicated by*  
 892 *three lines: dotted, normal and bold lines correspond to minimal, median and maximum*  
 893 *value of Cd concentration in soils (a and c) or ground-dwelling invertebrate richness in the*  
 894 *field (b and d), respectively. Mouse number is fixed to the median value for drawing the lines*  
 895 *(three mice in spring and two mice in autumn). The patterns of the relation between ground-*  
 896 *dwelling invertebrate richness in the diet and in the field are similar for flying invertebrate*  
 897 *richness. The patterns are similar for Pb and Zn concentrations in soils.*



898

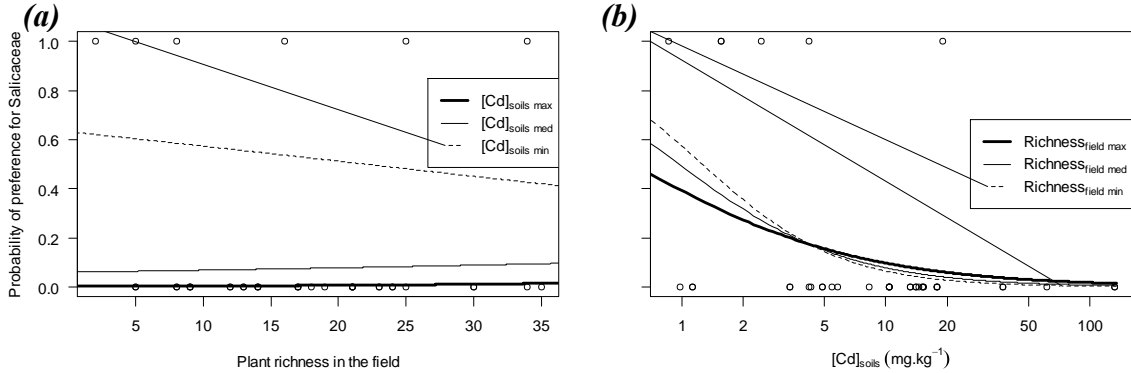
899  
900

901 Figure 3: Earthworm richness in the diet of the wood mouse along the gradient of Cd  
902 concentration in soils in spring (a) and in autumn (b). Each point represents a buffer and  
903 point size is proportional to the number of mice captured in a given buffer. Values predicted  
904 from our model are indicated by a line. Mouse number is fixed to the median value for  
905 drawing the line (three mice in spring and two mice in autumn). The patterns are similar for  
906 Pb and Zn concentrations in soils.



907  
908

909 *Figure 4: Predicted probabilities of preference for Salicaceae in spring along the gradient*  
 910 *of plant richness in the field (a) and Cd concentration in soils (b). Plotted points at*  
 911 *probability value 1 indicate buffers where preference for Salicaceae was observed, and*  
 912 *points at 0 indicate buffers where preference was not observed. Predicted probability form*  
 913 *our model is indicated by three lines: dotted, normal and bold lines correspond minimal,*  
 914 *median and maximum value of Cd concentrations in soils (a) or plant richness in the field*  
 915 *(b), respectively. Mouse number is fixed to the median value for drawing the lines (three*  
 916 *mice in spring and two mice in autumn). The patterns are similar for Pb and Zn*  
 917 *concentrations in soils.*



918