



HAL
open science

Selenium distribution in French forests: Influence of environmental conditions

Paulina Pisarek, Maïté Bueno, Yves Thiry, Manuel Nicolas, Hervé Gallard,
Isabelle Le Hécho

► To cite this version:

Paulina Pisarek, Maïté Bueno, Yves Thiry, Manuel Nicolas, Hervé Gallard, et al.. Selenium distribution in French forests: Influence of environmental conditions. *Science of the Total Environment*, 2021, 774, pp.144962. 10.1016/j.scitotenv.2021.144962 . hal-03162653

HAL Id: hal-03162653

<https://hal-univ-pau.archives-ouvertes.fr/hal-03162653>

Submitted on 10 Mar 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.



Distributed under a Creative Commons Attribution - NonCommercial| 4.0 International License

1 **Title**

2 **Selenium distribution in French forests: influence of**
3 **environmental conditions**

4

5 Paulina Pisarek^{1,2}, Maïté Bueno¹, Yves Thiry², Manuel Nicolas³, Hervé Gallard⁴, Isabelle Le
6 Hécho¹

7

8 ¹ CNRS/Univ. Pau & Pays de l'Adour/E2S UPPA, Institut des Sciences Analytiques et de
9 Physico-Chimie pour l'Environnement et les Matériaux (IPREM), UMR 5254, 64053 Pau,
10 France (*pisarek.paulina@univ-pau.fr) (maite.bueno@univ-pau.fr; isabelle.lehecho@univ-
11 pau.fr)

12 ² Andra, Research and Development Division, Parc de la Croix Blanche, 92298 Châtenay-
13 Malabry Cedex, France (yves.thiry@andra.fr)

14 ³ Office National des Forêts (ONF), Direction Forts et Risques Naturels, Département
15 Recherche, Développement, Innovation, Boulevard de Constance, 77300, Fontainebleau,
16 France (manuel.nicolas@onf.fr)

17 ⁴ IC2MP UMR 7285, Université de Poitiers, 86073 Poitiers Cedex 9, France
18 (herve.gallard@univ-poitiers.fr)

19

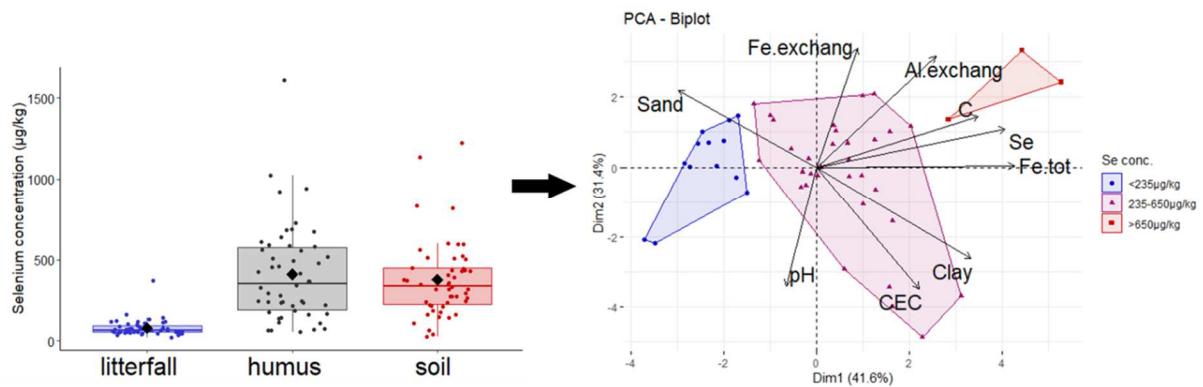
20 **Highlights**

- 21 • Statistical study of factors influencing Se distribution in 51 French forests
22 • Selenium concentration in litterfall increases with proximity to the ocean
23 • Low degradation rate of organic matter contribute to high Se stock in mor humus
24 • Aluminium, iron and organic matter content promote Se retention in soil
25

26 **Abstract**

27 Selenium is a trace element and an essential nutrient. Its long-lived radioisotope, selenium 79
28 is of potential radio-ecological concern in surface environment of deep geological repository
29 for high-level radioactive waste. In this study, the influence of environmental, climatic and
30 geochemical conditions on stable Se (as a surrogate of ⁷⁹Se) accumulation was statistically
31 assessed (PCA analysis, Kruskal-Wallis and Spearman tests) based on the analysis of its
32 concentration in litterfall, humus, and soil samples collected at 51 forest sites located in
33 France. Selenium concentrations were in the ranges: 22-369, 57-1608 and 25-1222 μg kg⁻¹
34 respectively in litterfall, humus, and soil. The proximity of the ocean and oceanic climate
35 promoted Se enrichment of litterfall, likely due to a significant reaction of wet deposits with
36 forest canopy. Se content was enhanced by humification (up to 6 times) suggesting that Se
37 concentrations in humus were affected by atmospheric inputs. Selenium stock in humus
38 decreased in the order of decreasing humus biomass and increasing turnover of organic
39 matter: mor > moder > mull. Positive correlations between Se content and geochemical
40 parameters such as organic carbon content, total Al and total Fe confirmed the important role
41 of organic matter (OM) and mineral Fe/Al oxides in Se retention in soils.

42 **Graphical abstract**



43

44 **Keywords**

45 ⁷⁹Se, Fe exchange, pH, CEC, humification, atmospheric deposition

46

47 **1. Introduction**

48 The radionuclide ^{79}Se is of potential radio-ecological concern at high-level nuclear waste
49 storage sites (Hjerpe and Broed, 2010) due to its long half-life (3.27×10^5 years) and its
50 capacity to be easily incorporated into natural biogeochemical cycle. Selenium is an essential
51 element in animal nutrition and metabolism with a major physiological role associated with
52 its presence in amino acids as selenocysteine (SeCys) and selenomethionine (SeMet)
53 (Rayman, 2000). Selenium is not essential for higher plants, although it is absorbed by plants
54 to various extent with some positive effects observed, especially for hyperaccumulators (El
55 Mehdawi and Pilon-Smits, 2012).

56 The global average Se content in the Earth' crust is estimated to be $50 \mu\text{g kg}^{-1}$ (Taylor and
57 McLennan, 1995), while average Se concentration in soil is $400 \mu\text{g kg}^{-1}$ (Natasha et al.,
58 2018). Selenium accumulation in surface soils depends on different Se inputs with variable
59 contributions. Geogenic sources appears to be the most important for Se-rich soils only, but it
60 does not explain the large-scale distribution of Se (Blazina et al., 2014). The ocean is an
61 important reservoir of Se, from where it can be volatilized and transported as sea spray
62 (Blazina et al., 2014; Sun et al., 2016; Mason et al., 2018). Therefore, atmospheric
63 deposition to nearby coast lines influences the Se content of vegetation and soil (Låg and
64 Steinnes, 1978), but there is little evidence of oceanic transfer to more remote terrestrial
65 regions (Blazina et al., 2014). Thus, another contribution to soil reserves appears to originate
66 from intermediate zones due to remobilization through volatilization, and subsequent wet and
67 dry deposition (Ross, 1985; Suess et al., 2019). In addition to natural sources, significant
68 anthropogenic Se contamination may come from dust, coal or petroleum combustion, as well
69 as agricultural sludges and fertilizers (Belon et al., 2012; Natasha et al., 2018)

70 The mobility of Se in soil is affected by various biological and geochemical factors such as
71 redox potential, pH, salinity, microbial activity, soil organic matter, and minerals such Fe and

72 Al oxides and clay (Dinh et al., 2017; Li et al. 2017). Among soil components, Fe/Al
73 minerals and organic matter (OM) play a pivotal role in Se persistence, but there is still no
74 agreement on the most important constituent (Coppin et al., 2009; Söderlund et al., 2016). It
75 is also due to complex, and yet unclear, Se-OM interaction mechanisms that have been
76 hypothesized to be mediated by mineral surfaces (Coppin et al., 2006) or by metal cations
77 within ternary complexes (Li et al., 2017; Martin et al., 2017; Tolu et al., 2014). In soil, Se
78 occurs in inorganic forms: Se(IV), Se(VI), Se(0), Se(-II) and in organic forms as complexes
79 with OM or organo-mineral colloids (Winkel et al., 2015). The Se content associated with the
80 organic fraction of soils ranges from 14 to 82% (Supriatin et al., 2015; Tolu et al., 2011;
81 Wang et al., 2012). In forest soils around $67 \pm 19\%$ of total Se is found in the organic
82 fraction, while it is $35 \pm 12\%$ in cultivated soils and $55 \pm 22\%$ in grasslands (Tolu et al.,
83 2014).

84 Limited data are available on Se content in forest compartments (Almahayni et al., 2017; Di
85 Tullo, 2015; Tolu, 2012; Tyler, 2005), especially in contrasted geogenic ecosystems. In this
86 study, particular attention was paid to forest ecosystems where Se inventories are expected to
87 characterize the long-term influence of natural inputs and element recycling processes. Here,
88 a comprehensive statistical study was proposed to determine the significance of eco-climatic
89 and geochemical conditions in Se accumulation and distribution at various forest sites. The
90 concentrations of selenium were determined in litterfall, humus, and soil samples from 51
91 different forest sites located in France. Fluxes and stocks of Se in forest compartments were
92 estimated, as well as time-scale involved in Se retention in humus and soil. Monitored sites
93 showed a wide variety in types of trees, climates, soils, and distance to coast. The sample
94 collection presented also the considerable benefit of complete characterization and regular
95 monitoring of forest sites provided by the ONF-RENECOFOR network (National Network
96 for the long term Monitoring of Forest Ecosystems).

97 **2. Material and methods**

98 **2.1. Study sites**

99 Samples of soil, humus, and litterfall were obtained from the RENECOFOR network handled
100 by ONF (French National Forest Office), which is part of the Pan-European ICP Forests
101 intensive (Level II) monitoring network. In order to benefit from available samples and data,
102 the same samples from 51 sites used by Redon et al. (2011) and Roulier et al. (2019) were
103 analyzed (Figure SI-1). These sites represent various forest types (dominance of 8 tree
104 species), climates (continental, transition, oceanic and mountainous), humus forms (mor,
105 moder, mull) and pedological properties (Table SI-1). The extensive data collection and sites
106 description was carried out as part of the RENECOFOR network (Brêthes , Urlich , 1997;
107 Probst et al., 2003; Redon et al., 2011; Roulier et al., 2019) (Table SI-2). Moreover Se
108 concentrations and fluxes in rainfall were measured by Roulier et al., (2020) in a subset of 27
109 sites. The maps with the location of the sampling sites was adapted in R Studio (R Core
110 Team, 2013; version 3.4.3). The coastal distance was estimated taking into account the
111 closest coast and the direction of the prevailing wind.

112 **2.2 Sampling procedure and sample treatment**

113 Mineral soil layers of systematic depth (0-10, 10-20, 20-40 cm) and humus horizons (litter,
114 fragmented, and humic layers) were sampled during the period 1993-1995 to measure their
115 dry mass and chemical concentrations. For each site and layer, five samples were taken for
116 chemical analyses from five 13.5 ×13.5 m² subplots distributed within the 0.5 ha central part
117 of the site, then oven-dried at 35°C and stored in the dark at room temperature (details in
118 Jonard et al., 2017). Litterfall was collected in ten traps of 0.5 m² placed at 1 m above forest
119 floor in every plots. Litterfall samples were harvested three to five times a year in the period
120 1998-1999, dried and homogenized. Soil, humus and litterfall composites were prepared by
121 mixing and grinding the individual samples in their respective mass proportions: soil layers

122 (0-10, 10-20, 20-40 cm), humus fractions (litter, fragmented and humic layers), and litterfall
123 (foliage, branches, fruits and 'rest material').

124 **2.3 Selenium extractions and analysis**

125 Approximately 0.25 g of soil, humus, or plant material were weighed in 50 ml DigiTUBE
126 (polypropylene). Soil was mixed with 0.5 mL of HCl and 1.5 mL of HNO₃, while humus and
127 litterfall were mixed with 2 mL of HNO₃, 0.5 mL of H₂O₂ and 3 mL of ultrapure water (18.2
128 MΩ.cm, Millipore Elix system). Samples were digested at 90 °C for 3 hours in hot block
129 digestion system (DigiPREP MS, SCP Science). Digested samples were diluted to 50 mL
130 with ultrapure water, filtered on 0.45 μm acetate membrane and stored at 4 °C until analysis.
131 Each sample was extracted and analyzed in triplicate except those in limited amount (detailed
132 in Table SI-3) and blanks were added in each digestion run.

133 Selenium concentration was determined by inductively coupled plasma mass spectrometry
134 (ICP-MS; 7500ce and 7900, Agilent Technologies, Tokyo, Japan) with collision/reaction cell
135 gas flow of 5 mL min⁻¹ H₂. Quantification was obtained by external calibrations in
136 reconstituted matrices (blanks). Instrumental detection limits for Se ranged between 0.4 - 8
137 μg kg⁻¹ depending on sample matrix. Typical analytical precision was lower than 10 %
138 (relative standard deviation, 15 replicates). Certified reference materials of soil DC 73030,
139 DC 73032 (National Analysis Center for Iron & Steel, Beijing, China) and vegetal material
140 ERM-CD281 (Rye grass, Community Bureau of Reference, Brussels, Belgium) were
141 analyzed in order to validate the accuracy of the extraction and analytical methods. Our
142 experimental values (191 ± 15 μg kg⁻¹ for DC 73030, 118 ± 9 μg kg⁻¹ for DC 73032, 24 ± 3
143 μg kg⁻¹ for ERM-CD281) agreed well with the certified values (200 ± 30 μg kg⁻¹ for DC
144 73030, 140 ± 20 μg kg⁻¹ for DC 73032, 23 ± 4 μg kg⁻¹ for ERM-CD281).

145 **2.4. Calculations**

146 Total soil mass (g ha^{-1}) was estimated as the sum of the masses of all soil layers (0-10, 10-20,
147 20-40 cm) that were calculated by multiplying the soil layer density by the layer thickness.
148 Selenium stock in the top 40 cm of soil (g ha^{-1}) was calculated by multiplying Se
149 concentration by soil mass. Selenium stocks in humus (g ha^{-1}) and annual litterfall (g ha^{-1}
150 yr^{-1}) were calculated by multiplying selenium concentration by the masses of corresponding
151 compartments. Information about soil bulk densities and masses of compartments were
152 provided by ONF and are given in Table SI-2.

153 Theoretical average residence time in forest soil (t_{resSe}) was calculated according to the
154 Equation 1:

155
$$t_{\text{resSe}} = \frac{\text{stock of Se in humus+mineral soil}}{\text{Annual wet atmospheric Se input}} \text{ (years)} \quad (1)$$

156
157 This calculation was based on steady-state assumption and on the fact that Se input from
158 weathering of the parent rock is negligible given the low average Se content of the Earth'
159 crust (Taylor and McLennan, 1995). Therefore, atmospheric deposition was considered the
160 only source of Se in the forest soil as proposed previously by Redon et al. (2011) and Bowley
161 (2013) for chlorine and iodine. Values of annual wet atmospheric Se used for calculation
162 were from Roulier et al. (2020).

163 Net Se accumulation rate in humus allowed estimating accumulation associated to vegetal
164 material decay (from litterfall to humus) and was calculated according to Equation 2 (Redon
165 et al., 2011):

166
$$\text{Se accumulation rate} = \frac{([\text{Se}]_{\text{humus}} - [\text{Se}]_{\text{litter}}) \times \text{DM}_{\text{humus}}}{t_{\text{resDM}}} \text{ (g ha}^{-1} \text{ yr}^{-1}) \quad (2)$$

167 where $[Se]_{\text{humus}}$ and $[Se]_{\text{litter}}$ are Se concentrations in humus and litterfall, DM_{humus} and t_{resDM}
168 are dry mass and the average dry matter residence time in the humus, respectively. t_{resDM} was
169 calculated according to Equation 3 (Redon et al., 2011):

$$170 \quad t_{\text{resDM}} = \frac{DM_{\text{humus}}}{LF \times (1 - \text{litter fraction mineralized})} \text{ (years)} \quad (3)$$

171 where LF is the annual litterfall dry mass. ‘litter fraction mineralized’ is the average mass
172 loss of litter due to mineralization before its total humification. For the calculation, litter
173 fraction mineralized values were taken from the study of Osono and Takeda (2005) on
174 different tree species (85% in pines, 70% in Douglas, 61% in spruce, 51% in silver fir, 65%
175 in beech and 43% in oak forests).

176

177 **2.5. Statistical analyses**

178 Statistical analyses were used to determine correlations between selenium concentrations or
179 stocks in soil, humus, and litterfall (response variables) with environmental, climatic, and
180 geochemical conditions.

181 Shapiro-Wilk test was first used to determine if selenium concentrations and stocks in soil,
182 humus and litterfall were normally distributed. As Se data were not normally distributed, a
183 Spearman rank correlation test and Kruskal-Wallis test were performed respectively for
184 numerical and categorical variables. Categorical variables comprised: forest type (coniferous,
185 deciduous), tree species (oak, beech, Douglas, pine, spruce, fir), climate (mountain,
186 continental, oceanic, transition), humus type (mor, mull, moder), rock type (sedimentary
187 rocks, igneous rocks, other), soil type (cambisol, planosol, podzoluvisol, luvisol, andosol,
188 podzol, letosol). Pairwise Wilcoxon test with Holm adjustment for p-values was applied in
189 order to establish the differences within sub-groups of categorical variables.

190 The distributions of selenium concentrations, stocks, and accumulation rates were represented
191 by boxplots for disparate categorical variables, with significant differences indicated by

192 letters. Principal component analysis (PCA) was performed and a biplot correlation circle
193 was drawn in order to visualise the relationships between soil numerical physico-chemical
194 parameters and selenium concentrations in soil and the contribution of individual sites into
195 the representation. Statistical analyses and boxplot representations were performed with R
196 Studio (R Core Team, 2013; version 3.4.3).

197

198 **3. Results and discussion**

199 **3.1 Selenium distribution in forest ecosystems**

200 Selenium concentrations in litterfall, humus, and soil samples of the forest sites are detailed
201 in Table SI-3. Selenium concentrations were in the ranges 22-369 $\mu\text{g kg}^{-1}$, 57-1608 $\mu\text{g kg}^{-1}$
202 and 25-1222 $\mu\text{g kg}^{-1}$ respectively for litterfall, humus and soil samples (Figure 1).

203 Selenium concentrations in humus were in the same range or higher than in soils while in
204 litterfall they were on average 6 times lower than in humus and soils (Kruskal-Wallis rank for
205 global test: $\chi^2 = 75.30$, p-value < 0.001). Selenium contents in litterfall were comparable to
206 values of 59 ± 11 (Di Tullo, 2015) and 112 $\mu\text{g kg}^{-1}$ (Tyler, 2005) previously reported for
207 beech leaf litter. A wide range of Se concentration in humus was also observed in the
208 literature. Di Tullo (2015) reported 151 $\mu\text{g kg}^{-1}$ for beech mull type humus and Tyler (2005)
209 the value of 650 $\mu\text{g kg}^{-1}$ for mor – humus formation of soil's O-horizon. Reimann et al.
210 (2015) also observed higher Se concentrations in the range 400-6300 $\mu\text{g kg}^{-1}$ in organic O-
211 horizon compared to 100-2700 $\mu\text{g kg}^{-1}$ in mineral C-horizon (20-80 cm depth) of Norwegian
212 forests. Selenium concentrations in soils (Figure 2) were coherent with average global values
213 from 10 to 2000 $\mu\text{g kg}^{-1}$ (Natasha et al., 2018) and values reported for European soils: 82 to
214 1119 $\mu\text{g kg}^{-1}$ in France (Tolu, 2012), 100 to 4000 $\mu\text{g kg}^{-1}$ in UK (Broadley et al., 2006) and
215 120 to 1970 $\mu\text{g kg}^{-1}$ in the Netherlands (Supriatin et al., 2015). Selenium concentrations in

216 humus were only moderately correlated with Se concentrations in litterfall (Spearman test, r_s
217 = 0.40, $p < 0.01$), whereas no correlation was found between Se concentrations in litterfall
218 and soil (p -value > 0.05). Moderate correlation between Se concentrations in humus and soil
219 was found (Spearman test, $r_s = 0.43$, p -value < 0.01).

220 The Se stock in 40 cm deep soil composites varied from 142 to 3702 g ha⁻¹, which is much
221 higher than Se reserves in humus (0.3 to 121 g ha⁻¹, Table SI-3). The mean annual Se flux
222 from litterfall to forest soil was 0.30 g ha⁻¹ yr⁻¹ with values ranging from 0.06 to 1.20 g ha⁻¹
223 yr⁻¹ (Table SI-3). Roulier et al (2020) reported annual Se fluxes in rainfall for a subset of the
224 same surveyed sites ($n = 27$) ranging from 0.25 to 2.01 g ha⁻¹ yr⁻¹ with a median value of
225 0.34 g ha⁻¹ yr⁻¹. Combining these two fluxes would result into apparent total annual Se
226 contribution to forest floor of 0.45 – 2.49 g ha⁻¹ yr⁻¹ (Table SI-3). The sum of these two
227 contributions might overestimate Se input to the soil given the interception (mean annual
228 interception fraction of the precipitation can reach 31% (Ulrich et al., 2002)) and potential
229 surface adsorption of a fraction of Se from rain by foliage before senescence. In addition to
230 wet deposition interception, it is also possible that Se in litterfall partly originated from dry
231 deposition or foliar or root uptake, different pathways not considered in the present budget.
232 On the other hand, Se volatilization that is common for many plants and microorganisms
233 (Chasteen and Bentley, 2003) might have compensated the impact of those different
234 processes in overall budget. In brief, the net accumulation of atmospheric Se by the foliage
235 remains uncertain and difficult to quantify as is the case for various trace elements.
236 Moreover, the results of correlation tests showed no direct dependence between Se
237 concentrations in humus and soil and the sum of Se contributions from rain and litterfall ($p >$
238 0.05, Table 1). This suggests that mobilization and immobilization processes of Se in forest
239 soils have a greater influence on the Se concentration of soils than the contributions of
240 rainfall and litterfall. When Se reaches the ground via litterfall, it can be released during

241 biomass decomposition, then volatilized, or leached to deeper forest soil layers where
242 sorption/desorption processes take place, or possibly taken up by tree roots and further
243 recycled. Further statistical analyses were performed in order to highlight the factors
244 affecting Se inventories in forest ecosystem.

245

246 **3.2 Environmental factors affecting Se concentration in litterfall**

247 Statistical analysis showed significant effect of distance from the coast, climate type, Se
248 concentration, and flux in rain on Se concentration in litterfall (Table 1). Furthermore, Se
249 concentration in litterfall was statistically homogeneous through forest types and tree species
250 (Kruskal-Wallis test, $p > 0.05$).

251 Slightly higher concentrations of Se in litterfall in oceanic climate compared to transition and
252 mountain climates (Figure 3.1) were confirmed by weak negative correlation between Se
253 concentration in litterfall and distance from the coast (Spearman test, $r_s = -0.35$, $p < 0.05$).

254 The ocean is an important source of atmospheric Se (Amouroux et al., 2001; Mason et al.,
255 2018; Wen and Carignan, 2007), reaching the ground or plant surface as dry and wet
256 depositions. Strong positive correlation was obtained between Se concentration in litterfall
257 and in rainfall when EPC08 site was excluded (Spearman test, $r_s = 0.73$, $p < 0.001$, $r^2 = 0.66$,
258 Figure 3.2). This site showed a very high Se concentration in litterfall ($369 \mu\text{g kg}^{-1}$)
259 compared to concentrations of the other sites (between 22 and $161 \mu\text{g kg}^{-1}$), suggesting local
260 contamination. The high concentrations of Se in soil ($1222 \mu\text{g kg}^{-1}$) and humus ($1608 \mu\text{g}$
261 kg^{-1}) at this site would be coherent with recurrent atmospheric emissions. The EPC08 site is
262 located in the Ardennes mountains, very close to the industrial regions of Western Europe
263 (Cumbers et al., 2006) whose economy had been based for years on metallurgical industry
264 requiring charcoal combustion. Nowadays, there are developed railway, car and battery
265 recycling industries, all of which can be a potential source of long-term local pollution.

266 Interestingly, the same site was also reported to be contaminated by As and atmospheric
267 deposits of Pb (Pauget et al., 2013; Probst et al., 2003).

268 In summary, Se concentration in litterfall was dependent from environmental conditions such
269 as climate, Se input from rainfall and proximity to the coast. Furthermore, anthropogenic
270 sources may contribute to Se deposition for some sites.

271

272 **3.3 Se accumulation in humus compartment**

273 As already discussed in Section 3.1, Se concentration in humus showed a wide range of
274 values from 57 to 1607 $\mu\text{g kg}^{-1}$ with median value of 353 $\mu\text{g kg}^{-1}$ and maximum value
275 corresponding to EPC08 potentially contaminated site. According to differences between
276 litterfall and humus, Se concentration was increased on average six times along the litter
277 decay continuum. Generally, Se concentrations in humus of coniferous forests were higher
278 than in deciduous forests (Figure 4.1, Table 1), despite homogenous concentrations of
279 selenium in litterfall of both forest types. Thus, humus under spruce and Douglas had
280 significantly higher Se content compared to oak (Figure SI-2, Table 1). It may result from
281 differences in the humification process, as deciduous leaves have a different composition
282 from conifer needles (e.g. lower lignin content), which generally leads to a higher labile mass
283 content and greater leaching losses in the early stages of litter decomposition (Prescott et al.,
284 2000). The calculated average residence times of humus were higher for Douglas fir, spruce
285 and pine (37.6 ± 27.7 , 54.2 ± 50.4 and 144.2 ± 113.5 years, Table SI-2) compared to oak and
286 beech (5.5 ± 5.0 and 12.4 ± 12.2 years Table SI-2), which reflects slower decomposition rates
287 of organic matter under conifers. Only humus from silver fir differed from the other
288 coniferous species by low masses ($20.3 \pm 15.5 \text{ t ha}^{-1}$) and low average residence times (10.8
289 ± 6.8 years). In the RENECOFOR network, forests of silver fir were usually associated to
290 mull humus formation on cambisol characterized by rapid litter decomposition.

291 As a consequence of different litter decay rates, Se concentration in humus was strongly
292 correlated with humus mass (Spearman rank test, $r_s = 0.68$, p-value < 0.001). This resulted in
293 lower humus Se stocks in oak, beech and silver fir forests (4 ± 6 , 8 ± 15 , 10 ± 9 g ha⁻¹)
294 compared to Douglas fir, spruce and pine (16 ± 11 , 47 ± 38 , 41 ± 38 g ha⁻¹). It was reflected
295 likewise in the dependence of the Se humus stock on the type of humus. Mean Se stock in
296 humus decreased in the order: mor > moder > mull according to decreasing humus biomass
297 and increasing organic matter degradation rate (Figure 4.2, Table 1). Selenium concentrations
298 in soils followed opposite order with significantly lower concentrations in soils under mor
299 compared to mull-humus formations (Figure 4.3) highlighting the importance of Se input
300 from humus decomposition to the underlying soil layers. Selenium can be mobilized in the
301 form of soluble (hydr)-oxyanions and/or organo-Se compounds (seleno-methionine (SeMet)
302 and methane seleninic acid (MeSeOOH) (Tolu et al., 2014)) and colloidal OM of plant or
303 microbial origin (Weng et al., 2011). Beside lower biomass and faster degradation, mull-
304 humus formation is also characterized by higher microbial activity, which can lead to the
305 release of volatile compounds such as H₂Se, DMSe (dimethyl selenide) and DMDSe
306 (dimethyl diselenide) into the atmosphere (Wessjohann et al., 2007).

307 The median Se accumulation rate in humus was 0.30 g ha⁻¹ yr⁻¹ with no clear distinction
308 between coniferous and deciduous tree species (Table SI-3). The highest accumulation rate,
309 1.57 g ha⁻¹ yr⁻¹, was for the EPC08 site. For this site, a long lasting dry deposition from
310 anthropogenic contamination may affect accumulation rate value. Generally, Se accumulation
311 rates were positive, except for one beech forest site (HET55: -0.02 ± 0.01 g ha⁻¹ yr⁻¹) for
312 which Se concentrations in humus and litterfall were similar (57 ± 6 and 73 ± 8 , respectively,
313 Table SI-3). Negative accumulation rates were reported for chlorine in beech and oak forests,
314 while they were positive for chlorine in Douglas, fir, spruce, and pine forests (Redon et al.,
315 2011), and for iodine in all tree species (Roulier et al., 2019). Negative accumulation rates

316 were interpreted as a result of high humus decomposition rate and leaching, whereas positive
317 accumulation rates reflect Se enrichment during the process of litterfall transformation into
318 humus. The average rate of Se accumulation in humus ($0.39 \text{ g ha}^{-1} \text{ yr}^{-1}$) was similar to the
319 average Se flux from litterfall ($0.30 \text{ g ha}^{-1} \text{ yr}^{-1}$), and the average rainfall Se flux (0.44 g ha^{-1}
320 yr^{-1}). This suggests a partial humus enrichment by the capture of soluble Se from rain
321 through sorptive accumulation. Beside the retention, Se of atmospheric origin is subject to
322 losses by drainage and volatilization, but the contribution of each pathway remains unclear
323 and needs further investigation.

324 To conclude, Se inventories in humus were mostly controlled by the litter decay process and
325 the rate of organic matter decomposition and not directly by the input of Se to the forest floor.

326

327 **3.4 Environmental variables influencing Se content in soil**

328 The soil compartment was shown to be the main reservoir of Se. Tree species, climate and
329 rock type were the most discriminating categorical variables (Paragraph 3.3) to explain the
330 variation in Se content in French forest soils. Although the ocean is an important reservoir of
331 Se on a global scale, no direct correlation was found between Se concentration in soil and
332 distance from the coast. Decreasing Se content was observed with the distance from the
333 ocean in forest soil in Norway (Låg and Steinnes, 1978). Blazina et al. (2014) showed that
334 over the last 6.8 million years the monsoonal climate had likely dominant role on
335 atmospheric Se deposition and soil concentration in Chinese Loess Plateau. Winkel et al.
336 (2015) highlighted the need for further research to determine the extent to which wet
337 deposition influences soil Se levels on a large scale, although Se generally measured in wet
338 deposition is not location-dependent. In our study higher Se concentrations in soil were only
339 observed for the mountain climate compared to continental climate (Figure 5.1, Table 2). It
340 may be due to reduced microbial activity and lower degradation rate of OM in colder climate

341 or specific soil composition in the mountain climate. The former would have minor impact,
342 as no dependence was observed between Se concentrations in humus and climate. The latter
343 would be more important as mountainous soils had elevated total Fe, total Al and organic C
344 contents (Figures SI 5-7). Another co-correlation was found between climate and tree species,
345 as Se-poor transition forest soils were dominated by pines and oaks, while Se-rich
346 mountainous forest soils by Douglas. Summing up, the co-correlations between Fe, Al, C
347 contents in soil, climate and tree species do not permit to determine which of these factors
348 plays a dominant role in Se immobilization in soil.

349 No dependence was observed between Se concentrations in soil and soil type ($p > 0.05$, Table
350 2), which may be due to large variability in soil composition within a soil type. Concerning
351 the effect of bedrock material, soils derived from sedimentary rocks were in general poorer in
352 Se than those from igneous rocks (Figure 5.2, Table 1), which is contrary to some statements
353 in the literature (Sharma et al., 2015). Average Se level in igneous rocks was reported as 0.35
354 mg kg^{-1} , while the content in sedimentary rocks varies significantly within classification
355 groups: 0.05 mg kg^{-1} in sandstone, 0.08 mg kg^{-1} in limestone and the highest values for rocks
356 with high organic carbon content: 0.6 mg kg^{-1} for shale and 300 mg kg^{-1} for coal (Natasha et
357 al., 2018). Thus, Se concentrations measured in 0-40 cm soil depth may not reflect the
358 character of rock, especially for low-Se parent materials. Additionally, as analyzed soils were
359 from shallow soil layer (≤ 40 cm), Se input from litterfall and atmospheric depositions masks
360 the influence of soil parent material weathering.

361

362 **3.5 Effect of soil composition on Se content**

363 A correlation matrix was first constructed with all numerical variables describing soil
364 composition (Table SI-4). The variables that were well correlated (Spearman rank test $r_s >$
365 0.6) with soil Se concentrations were total Al, total Fe, total N, organic C, and clay fraction.

366 Moreover, there were also moderate correlations with soil exchangeable Al concentration,
367 sand and silt fractions, and cation exchange capacity (CEC). PCA analysis was then used to
368 depict the effect of soil composition on Se concentrations of individual soils. Sites EPC63
369 and EPC08 were excluded in PCA analysis because they showed peculiar characteristics, as
370 highlighted by their position with respect to linear correlations between Se concentration and
371 total Fe/Al and C_{org} (Figure SI-6, $r^2 = 0.75$ and $r^2 = 0.62$, respectively). As previously
372 discussed (Section 3.2), EPC08 was identified as a potentially contaminated site. The soil at
373 EPC63 site was the only representative of Andosol, which is rare for French geographical
374 location (about 1% in France) (Quantin, 2004).

375 The contribution of each variable to the first three principal components of PCA is presented
376 in Table 2. Further, the number of variables was reduced by removing co-correlated
377 parameters. Therefore, organic C was selected as the sole representative of soil OM
378 (elimination of total N) and total Fe as the sole representative of Fe/Al minerals (elimination
379 of total Al). Mn exchangeable was excluded as it contributed little to Dim 1 and Dim 2
380 components (Table 2) and any significant effect (p -value > 0.05) of soil exchangeable Mn on
381 Se concentration was observed. It was not surprising as selenite is more readily released to
382 liquid phase by anion-exchange mechanism from Mn oxides compared to goethite (Fe
383 oxides) (Saeki and Matsumoto, 1994).

384 The first two principal components explained 73.0% of the variation (Dim1 41.6 % and Dim2
385 31.4%). Selenium, C_{org} and Fe concentrations in soil were characterized by strongly positive
386 coordinates on the Dim 1 axis (Figure 6). Moreover, PCA eigenvalues showed positive loads
387 in the Dim1 for clay, Al exchangeable, CEC and Fe exchangeable variables. The second
388 component was well described by pH and Fe exchangeable variables that were logically anti-
389 correlated. Similar anti-correlation was observed between sand and clay fractions.

390 Soil samples were discriminated in three clusters of different Se concentrations: <235 $\mu\text{g kg}^{-1}$
391 kg^{-1} , 235-650 $\mu\text{g kg}^{-1}$, >650 $\mu\text{g kg}^{-1}$. Located on negative Dim 1, the first cluster with the
392 lowest Se concentrations < 235 $\mu\text{g kg}^{-1}$ including 13 sites was characterized by soils with
393 high contribution of sand and low C_{org} and Fe/Al content. This cluster includes mainly
394 arenosols, podzols, and planosols in oceanic and transition climates with maritime pine and
395 oak as the main tree species. In this cluster, sites PM17 and PM85 are two arenosols close to
396 the ocean (≤ 2 km) with the lowest Se concentrations (25 $\mu\text{g kg}^{-1}$ and 39 $\mu\text{g kg}^{-1}$,
397 respectively). Their position in the biplot is explained by their very high content of sand (99%
398 and 96% respectively), high pH values (8.7 and 8.5 respectively), and low OM content (3.85
399 and 5.55 g kg^{-1} respectively). These results confirmed the importance of OM and Al/Fe
400 content in Se retention and the fact that sand fraction is a weak Se carrier (Chi et al., 2019;
401 Coppin et al., 2006; Saeki and Matsumoto, 1994).

402 At the opposite on the biplot graph, the three soils with the highest Se concentrations (> 650
403 $\mu\text{g kg}^{-1}$; HET30, DOU71 and DOU65 sites) had positive coordinates on both dimensions
404 axes associated with Se, C_{org} , total Fe and Al exchangeable variables. These soils are acidic
405 podzols (HET30 and DOU 71) or cambisol (DOU65) in mountain climate with very high C_{org}
406 (67 – 85 g kg^{-1}), Fe (8.7 – 14.3 g kg^{-1}) and Al (7.1 – 12.0 g kg^{-1}) contents. In our study the
407 concentrations of organic C (and total N) were strongly correlated with those of total Fe and
408 Al (Table SI-6). Strong correlation between Se content and organic C concentration in soil is
409 related to the ability of OM to interact with Se. Aluminium and Fe oxides remain protonated
410 in a broad pH range and have relatively large surface area with densely arrayed functional
411 groups. These functional groups can also promote the formation of inner and outer sphere
412 complexes with Se (Peak, 2006; Söderlund et al., 2016). Like in our study, hot spots of Se in
413 soils were identified as co-located with Fe and Al contents suggesting possible indirect
414 sorption on metals associated within OM surface or matrix (Coppin et al., 2009).

415 Soils with medium Se concentrations (235 - 650 $\mu\text{g kg}^{-1}$) were gathered in the third cluster
416 (32 sites for a total of 49). This cluster contains all types of soils (except the two arenosols
417 discussed previously), tree species and climates. Within this cluster, a sub-group of 4 sites
418 (SP25, SP11, SP05 and HET25) for three soil types (luvisol, cambisol and leptosol) was
419 characterized by a strong contribution of clay fraction (35 – 50 %) and neutral pH (5.76 –
420 7.21). With moderate C_{org} (23 – 31 g kg^{-1}) and Al/Fe content (from 1.9 to 5.1 g Al kg^{-1} and
421 4.2 to 9.7 g Fe kg^{-1}), they also exhibited moderately high Se concentrations (371 – 500 μg
422 kg^{-1}).

423 Summarizing all of the above, our results confirm the strong relationships between Se and
424 organic matter and Al/Fe constituents. However, it was not possible to clarify whether Fe/Al
425 oxides or OM have a greater influence on Se sorption, as these variables were highly
426 correlated and may act synergistically on Se retention.

427 The calculated selenium residence times in forest soils (t_{resSe}) were between 617 and 8406
428 years (median value 2735 years for $n = 27$ sites), which is higher than retention times of
429 iodine (419 - 1756 years) and chlorine (3 - 67 years) for the same sites (Redon et al., 2011;
430 Roulier et al., 2019). That pattern is in agreement with other results showing higher
431 desorption distribution coefficients K_D in woodland for Se (41 - 1198 L kg^{-1} (Almahayni et
432 al., 2017; Tolu et al., 2014)) compared to I (4 - 281 L kg^{-1}) as well as higher sorption
433 coefficient K_D values for Se (4 - 2100 L kg^{-1}) than for I (0.01 - 580 L kg^{-1}) and Cl (0.04 - 1.2
434 L kg^{-1}) in all types of soils (IAEA, 2010). Summing up, radionuclide ^{79}Se has a highest
435 potential to persist in soil compartment compared to the other radionuclides ^{129}I and ^{36}Cl . The
436 broad range of t_{resSe} is a result of combinations of local environmental factors including input
437 to soil, organic matter turnover, and soil properties.

438 **4. Conclusions**

439 In this study, the influence of environmental and geochemical conditions on Se accumulation
440 in forest soil was analyzed. Selenium concentration in litterfall was linearly dependent on its
441 concentration in rainfall with higher values near the coast, i.e. for oceanic climate. Deviation
442 from correlation was observed for one site with very high concentration in litterfall (369
443 $\mu\text{g}/\text{kg}$) and was attributed to anthropogenic activities. The annual amount of selenium cycling
444 through litterfall was minor compared to Se stock in forest soils and, no direct correlation was
445 found between the two. The higher Se concentrations of recalcitrant mor humus compared to
446 easily degradable mull humus highlighted that organic matter degradation rate likely controls
447 Se content in humus and more generally in soils. In forest soils, the content of organic matter
448 and Fe/Al oxides had pivotal role among all soil constituents in long-term Se retention. Thus,
449 acidic soils rich in organic matter and Fe/Al oxides under Douglas fir and beech forests in
450 mountain climate are the soil properties and environmental conditions that promote the
451 persistence of Se in forest soils.

452

453 **Supplementary information**

454 Additional information as noted in the text.

455 **Acknowledgements**

456 The authors would like to thank all technical support from the RENECOFOR-ONF network,
457 which provided the collection of samples and sites data. We wish to acknowledge M. Roulier
458 for her guidance with statistical program.

459 Funding: This research was financially supported by the French National Radioactive Waste
460 Management Agency (Andra).

461 **References**

- 462 Almahayni, T., Bailey, E., Crout, N.M.J., Shaw, G., 2017. Effects of incubation time and
 463 filtration method on Kd of indigenous selenium and iodine in temperate soils. *J.*
 464 *Environ. Radioact.* 177, 84–90. <https://doi.org/10.1016/j.jenvrad.2017.06.004>
- 465 Amouroux, D., Liss, P.S., Tessier, E., Hamren-Larsson, M., Donard, O.F.X., 2001. Role of
 466 oceans as biogenic sources of selenium. *Earth Planet. Sci. Lett.* 189, 277–283.
 467 [https://doi.org/10.1016/S0012-821X\(01\)00370-3](https://doi.org/10.1016/S0012-821X(01)00370-3)
- 468 Belon, E., Boisson, M., Deportes, I.Z., Eglin, T.K., Feix, I., Bispo, A.O., Galsomies, L.,
 469 Leblond, S., Guellier, C.R., 2012. An inventory of trace elements inputs to French
 470 agricultural soils. *Sci. Total Environ.* 439, 87–95.
 471 <https://doi.org/10.1016/j.scitotenv.2012.09.011>
- 472 Blazina, T., Sun, Y., Voegelin, A., Lenz, M., Berg, M., Winkel, L.H.E., 2014. Terrestrial
 473 selenium distribution in China is potentially linked to monsoonal climate. *Nat.*
 474 *Commun.* 5, 1–7. <https://doi.org/10.1038/ncomms5717>
- 475 Bowley, H.E., 2013. Iodine dynamics in the terrestrial environment PhD Thesis. University
 476 of Nottingham.
- 477 Brêthes A., Ulrich E., 1997. Caractéristiques pédologiques des 102 peuplement du réseau. -
 478 Office national de forêts, Dépt des recherches Techniques.
- 479 Broadley, M.R., White, P.J., Bryson, R.J., Meacham, M.C., Bowen, H.C., Johnson, S.E.,
 480 Hawkesford, M.J., McGrath, S.P., Zhao, F.-J., Breward, N., Harriman, M., Tucker,
 481 M., 2006. Biofortification of UK food crops with selenium. *Proc. Nutr. Soc.* 65, 169–
 482 181. <https://doi.org/10.1079/pns2006490>
- 483 Chasteen, T.G., Bentley, R., 2003. Biomethylation of selenium and tellurium:
 484 microorganisms and plants. *Chem. Rev.* 103, 1–25. <https://doi.org/10.1021/cr010210+>
- 485 Chi, F.Q., Kuang, E.J., Zhang, J.M., Su, Q.R., Chen, X.L., Zhang, Y.W., Liu, Y.D., Kuang,
 486 E.J., Zhang, J.M., Su, Q.R., Chen, X.L., Zhang, Y.W., Liu, Y.D., 2019. Fractionation
 487 and distribution of soil selenium and effects of soil properties in Heilongjiang.
 488 *Selenium Research for Environment and Human Health: Perspectives, Technologies*
 489 *and Advancements.* <https://doi.org/10.1201/9780429423482-9>
- 490 Coppin, F., Chabroulet, C., Martin-Garin, A., 2009. Selenite interactions with some
 491 particulate organic and mineral fractions isolated from a natural grassland soil. *Eur. J.*
 492 *Soil Sci.* 60, 369–376. <https://doi.org/10.1111/j.1365-2389.2009.01127.x>
- 493 Coppin, F., Chabroulet, C., Martin-Garin, A., Balesdent, J., Gaudet, J.P., 2006.
 494 Methodological approach to assess the effect of soil ageing on selenium behaviour:
 495 first results concerning mobility and solid fractionation of selenium. *Biol. Fertil. Soils*
 496 42, 379–386. <https://doi.org/10.1007/s00374-006-0080-y>
- 497 Cumbers, A., Birch, K., MacKinnon, D., 2006. Revisiting the Old Industrial Region:
 498 Adaptation and Adjustment in an Integrating Europe. CPPR Working Paper 1.
 499 University of Glasgow.
- 500 Di Tullo, P., 2015. Dynamique du cycle biogéochimique du sélénium en écosystèmes
 501 terrestres : rétention et réactivité dans le sol, rôle de la végétation (thesis). Pau.
- 502 Dinh, Q.T., Li, Z., Tran, T.A.T., Wang, D., Liang, D., 2017b. Role of organic acids on the
 503 bioavailability of selenium in soil: A review. *Chemosphere* 184, 618–635.
 504 <https://doi.org/10.1016/j.chemosphere.2017.06.034>
- 505 El Mehdawi, A.F., Pilon-Smits, E. a. H., 2012. Ecological aspects of plant selenium
 506 hyperaccumulation. *Plant. Biol.* 14, 1–10. [https://doi.org/10.1111/j.1438-](https://doi.org/10.1111/j.1438-8677.2011.00535.x)
 507 [8677.2011.00535.x](https://doi.org/10.1111/j.1438-8677.2011.00535.x)
- 508 Hjerpe, T., Broed, R., 2010. Radionuclide transport and dose assessment modelling in
 509 biosphere assessment 2009 (No. POSIVA-WR--10-79). Posiva Oy.

- 510 IAEA, 2010. Handbook of Parameter Values for the Prediction of Radionuclide Transfer in
511 Terrestrial and Freshwater Environment.
- 512 Jonard, M., Nicolas, M., Coomes, D.A., Caignet, I., Saenger, A., Ponette, Q., 2017. Forest
513 soils in France are sequestering substantial amounts of carbon. *Sci. Total Environ.*
514 574, 616–628. <https://doi.org/10.1016/j.scitotenv.2016.09.028>
- 515 Låg, J., Steinnes, E., 1978. Regional distribution of selenium and arsenic in humus layers of
516 Norwegian forest soils. *Geoderma* 20, 3–14. [https://doi.org/10.1016/0016-](https://doi.org/10.1016/0016-7061(78)90045-9)
517 7061(78)90045-9
- 518 Li, Z., Liang, D., Peng, Q., Cui, Z., Huang, J., Lin, Z., 2017. Interaction between selenium
519 and soil organic matter and its impact on soil selenium bioavailability: A review.
520 *Geoderma* 295, 69–79. <https://doi.org/10.1016/j.geoderma.2017.02.019>
- 521 Martin, D.P., Seiter, J.M., Lafferty, B.J., Bednar, A.J., 2017. Exploring the ability of cations
522 to facilitate binding between inorganic oxyanions and humic acid. *Chemosphere* 166,
523 192–196. <https://doi.org/10.1016/j.chemosphere.2016.09.084>
- 524 Mason, R.P., Soerensen, A.L., DiMento, B.P., Balcom, P.H., 2018. The global marine
525 selenium cycle: insights from measurements and modeling. *Global Biogeochemical*
526 *Cycles* 32, 1720–1737. <https://doi.org/10.1029/2018GB006029>
- 527 Natasha, N., Shahid, M., Niazi, N.K., Khalid, S., Murtaza, B., Bibi, I., Rashid, M.I., 2018. A
528 critical review of selenium biogeochemical behavior in soil-plant system with an
529 inference to human health. *Environ. Pollut.* 234, 915–934.
530 <https://doi.org/10.1016/j.envpol.2017.12.019>
- 531 Osono, T., Takeda, H., 2005. Limit values for decomposition and convergence process of
532 lignocellulose fraction in decomposing leaf litter of 14 tree species in a cool temperate
533 forest. *Ecol. Res.* 20, 51–58. <https://doi.org/10.1007/s11284-004-0011-z>
- 534 Pauget, B., Gimbert, F., Coeurdassier, M., Crini, N., Pérès, G., Faure, O., Douay, F., Hitmi,
535 A., Beguiristain, T., Alaphilippe, A., Guernion, M., Houot, S., Legras, M., Vian, J.-F.,
536 Hedde, M., Bispo, A., Grand, C., de Vaufléury, A., 2013. Ranking field site
537 management priorities according to their metal transfer to snails. *Ecol. Indic.* 29, 445–
538 454. <https://doi.org/10.1016/j.ecolind.2013.01.012>
- 539 Peak, D., 2006. Adsorption mechanisms of selenium oxyanions at the aluminum oxide/water
540 interface. *J. Colloid Interface Sci.* 303, 337–345.
541 <https://doi.org/10.1016/j.jcis.2006.08.014>
- 542 Prescott, C.E., Zabek, L.M., Staley, C.L., Kabzems, R., 2000. Decomposition of broadleaf
543 and needle litter in forests of British Columbia: influences of litter type, forest type,
544 and litter mixtures. *Can. J. For. Res.* 30, 1742–1750. <https://doi.org/10.1139/x00-097>
- 545 Probst A., Hernandez L., Fevrier-Vauleon C., Prudent P., Probst J., Party J., 2003. Eléments
546 traces métalliques dans les sols des écosystèmes forestiers français: distribution et
547 facteurs de contrôle -utilisation du réseau RENECOFOR. Office National des
548 Forêts, Direction Technique.
- 549 Quantin, P., 2004. Volcanic soils of France. *CATENA, Volcanic Soil Resources: Occurrence,*
550 *Development and Properties* 56, 95–109. <https://doi.org/10.1016/j.catena.2003.10.019>
- 551 Rayman, M.P., 2000. The importance of selenium to human health. *Lancet* 356, 233–241.
552 [https://doi.org/10.1016/S0140-6736\(00\)02490-9](https://doi.org/10.1016/S0140-6736(00)02490-9)
- 553 Redon, P.-O., Abdelouas, A., Bastviken, D., Cecchini, S., Nicolas, M., Thiry, Y., 2011.
554 Chloride and organic chlorine in forest soils: storage, residence times, and influence
555 of ecological conditions. *Environ. Sci. Technol.* 45, 7202–7208.
556 <https://doi.org/10.1021/es2011918>
- 557 Reimann, C., Englmaier, P., Fabian, K., Gough, L., Lamothe, P., Smith, D., 2015.
558 Biogeochemical plant–soil interaction: Variable element composition in leaves of four

559 plant species collected along a south–north transect at the southern tip of Norway. *Sci.*
560 *Total Environ.* 506–507, 480–495. <https://doi.org/10.1016/j.scitotenv.2014.10.079>

561 Ross, H.B., 1985. An atmospheric selenium budget for the region 30° N to 90° N. *Tellus B*
562 37, 78–90. <https://doi.org/10.3402/tellusb.v37i2.14999>

563 Roulier, M., Bueno, M., Coppin, F., Nicolas, M., Thiry, Y., Rigal, F., Le Hécho, I., Pannier,
564 F., 2020. Atmospheric iodine, selenium and caesium depositions in France: I. Spatial
565 and seasonal variations. *Chemosphere* 128971.
566 <https://doi.org/10.1016/j.chemosphere.2020.128971>

567 Roulier, M., Coppin, F., Bueno, M., Nicolas, M., Thiry, Y., Della Vedova, C., Février, L.,
568 Pannier, F., Le Hécho, I., 2019. Iodine budget in forest soils: Influence of
569 environmental conditions and soil physicochemical properties. *Chemosphere* 224, 20–
570 28. <https://doi.org/10.1016/j.chemosphere.2019.02.060>

571 Saeki, K., Matsumoto, S., 1994. Selenite adsorption by a variety of oxides. *Commun. Soil*
572 *Sci. Plant Anal.* 25, 2147–2158. <https://doi.org/10.1080/00103629409369178>

573 Sharma, V.K., McDonald, T.J., Sohn, M., Anquandah, G.A.K., Pettine, M., Zboril, R., 2015.
574 Biogeochemistry of selenium. A review. *Environ. Chem. Lett.* 13, 49–58.
575 <https://doi.org/10.1007/s10311-014-0487-x>

576 Söderlund, M., Virkanen, J., Holgersson, S., Lehto, J., 2016. Sorption and speciation of
577 selenium in boreal forest soil. *J. Environ. Radioact.* 164, 220–231.
578 <https://doi.org/10.1016/j.jenvrad.2016.08.006>

579 Suess, E., Aemisegger, F., Sonke, J.E., Sprenger, M., Wernli, H., Winkel, L.H.E., 2019.
580 Marine versus Continental Sources of Iodine and Selenium in Rainfall at Two
581 European High-Altitude Locations. *Environ. Sci. Technol.* 53, 1905–1917.
582 <https://doi.org/10.1021/acs.est.8b05533>

583 Sun, G.-X., Meharg, A.A., Li, G., Chen, Z., Yang, L., Chen, S.-C., Zhu, Y.-G., 2016.
584 Distribution of soil selenium in China is potentially controlled by deposition and
585 volatilization? *Scientific Reports* 6. <https://doi.org/10.1038/srep20953>

586 Supriatin, S., Weng, L., Comans, R.N.J., 2015. Selenium speciation and extractability in
587 Dutch agricultural soils. *Sci. Total Environ.* 532, 368–382.
588 <https://doi.org/10.1016/j.scitotenv.2015.06.005>

589 Taylor, S.R., McLennan, S.M., 1995. The geochemical evolution of the continental crust.
590 *Rev. of Geophys.* 33, 241–265. <https://doi.org/10.1029/95RG00262>

591 Tolu, J., 2012. Spéciation et mobilité du sélénium présent dans les sols à l'état de traces :
592 contribution aux prévisions à long terme (thesis). Pau.

593 Tolu, J., Le Hécho, I., Bueno, M., Thiry, Y., Potin-Gautier, M., 2011. Selenium speciation
594 analysis at trace level in soils. *Anal. Chim. Acta* 684, 126–133.
595 <https://doi.org/10.1016/j.aca.2010.10.044>

596 Tolu, J., Thiry, Y., Bueno, M., Jolivet, C., Potin-Gautier, M., Le Hécho, I., 2014. Distribution
597 and speciation of ambient selenium in contrasted soils, from mineral to organic rich.
598 *Sci. Total Environ.* 479–480, 93–101. <https://doi.org/10.1016/j.scitotenv.2014.01.079>

599 Tyler, G., 2005. Changes in the concentrations of major, minor and rare-earth elements
600 during leaf senescence and decomposition in a *Fagus sylvatica* forest. *For. Ecol.*
601 *Manag.* 206, 167–177. <https://doi.org/10.1016/j.foreco.2004.10.065>

602 Ulrich, E., Coddeville, P., Lanier, M., 2002. Retombées atmosphériques humides en France
603 entre 1993 et 1998. ADEME, Paris.

604 Wang, S., Liang, D., Wang, D., Wei, W., Fu, D., Lin, Z., 2012. Selenium fractionation and
605 speciation in agriculture soils and accumulation in corn (*Zea mays* L.) under field
606 conditions in Shaanxi Province, China. *Sci. Total Environ.* 427–428, 159–164.
607 <https://doi.org/10.1016/j.scitotenv.2012.03.091>

- 608 Wen, H., Carignan, J., 2007. Reviews on atmospheric selenium: Emissions, speciation and
609 fate. *Atmos. Environ.* 41, 7151–7165. <https://doi.org/10.1016/j.atmosenv.2007.07.035>
- 610 Weng, L., Vega, F.A., Supriatin, S., Bussink, W., Riemsdijk, W.H.V., 2011. Speciation of Se
611 and DOC in Soil Solution and Their Relation to Se Bioavailability. *Environ. Sci.*
612 *Technol.* 45, 262–267. <https://doi.org/10.1021/es1016119>
- 613 Wessjohann, L.A., Schneider, A., Abbas, M., Brandt, W., 2007. Selenium in chemistry and
614 biochemistry in comparison to sulfur. *Biol. Chem.* 388, 997–1006.
615 <https://doi.org/10.1515/BC.2007.138>
- 616 Winkel, L.H.E., Vriens, B., Jones, G.D., Schneider, L.S., Pilon-Smits, E., Bañuelos, G.S.,
617 2015. Selenium cycling across soil-plant-atmosphere interfaces: A critical review.
618 *Nutrients* 7, 4199–4239. <https://doi.org/10.3390/nu7064199>
- 619

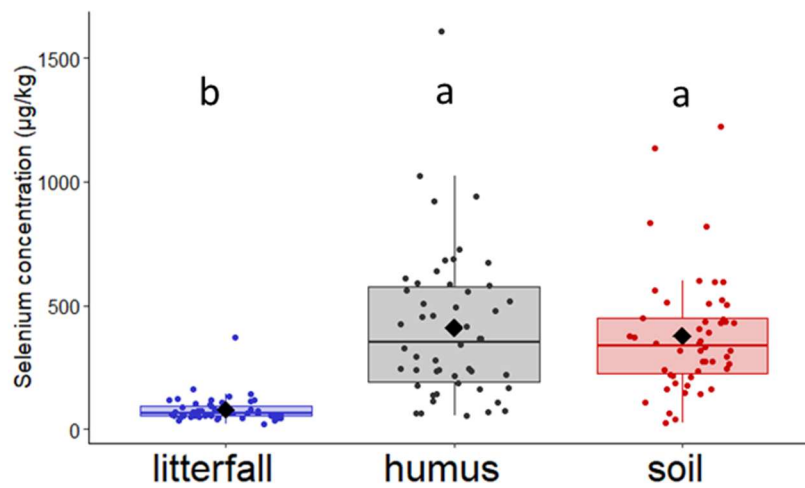


Figure 1. Selenium concentrations in litterfall, humus and soils. Box plots show the first quartile (Q_1), median and third quartile (Q_3) of the data. Black diamond symbols represent average Se concentrations in groups. Lower and upper whiskers extend from the hinges by $Q_1 - 1.5 \times IQR$ (IQR - interquartile range) and $Q_3 + 1.5 \times IQR$, respectively. Dots represent corresponding data. The different letters above the boxes correspond to significant differences between the groups according to pairwise Wilcoxon test, with Holm adjustment for p-value.

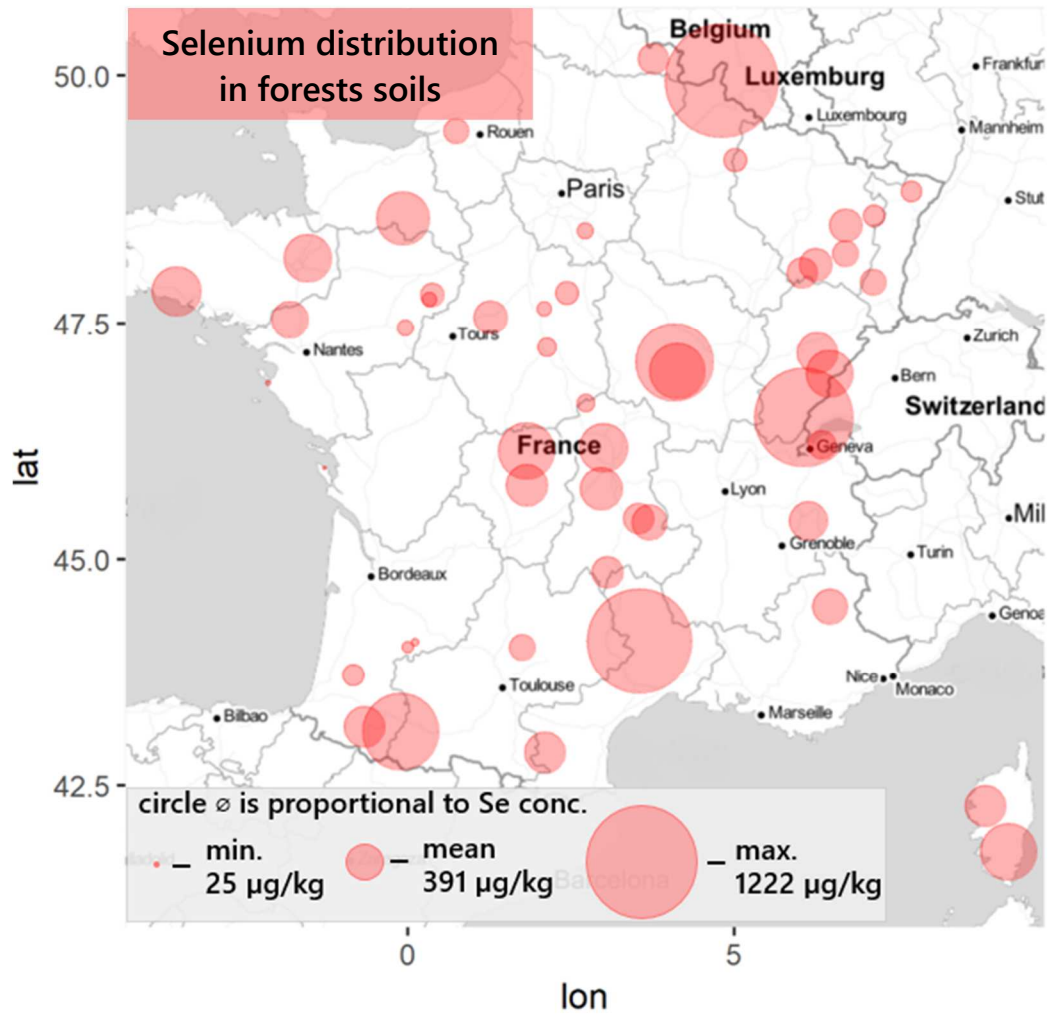


Figure 2 Selenium concentrations in French forest soils (mean value of top 40 cm soil). Diameter of red circle is proportional to Se concentration (minimum: 25 $\mu\text{g kg}^{-1}$; maximum: 1222 $\mu\text{g kg}^{-1}$).

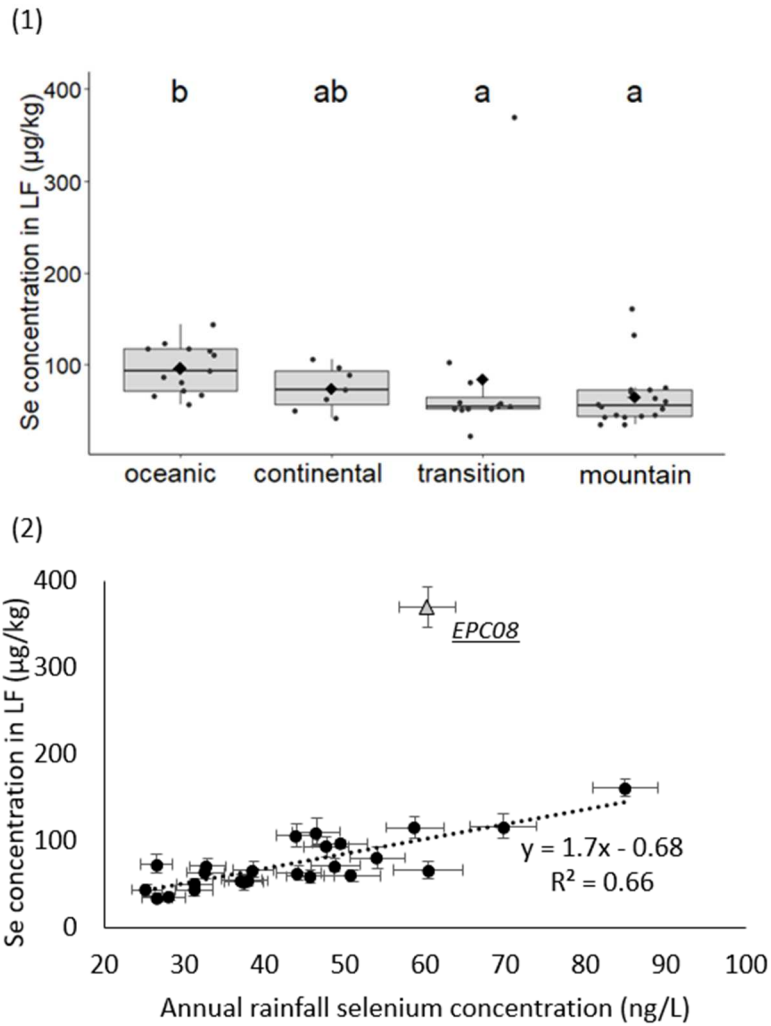


Figure 3 Se concentration in litterfall according to 1) climate; 2) annual volume-weighted mean Se concentration in rainfall. In Figure 3.1 box plots show the first quartile (Q_1), median and third quartile (Q_3) of the data. Black diamond symbols represent average Se concentrations in groups. Lower and upper whiskers extend from the hinges by $Q_1 - 1.5 \times \text{IQR}$ (IQR - interquartile range) and $Q_3 + 1.5 \times \text{IQR}$, respectively. Dots represent corresponding data. The different letters above the boxes correspond to significant differences between the groups according to pairwise Wilcoxon test, with Holm adjustment for p-value.

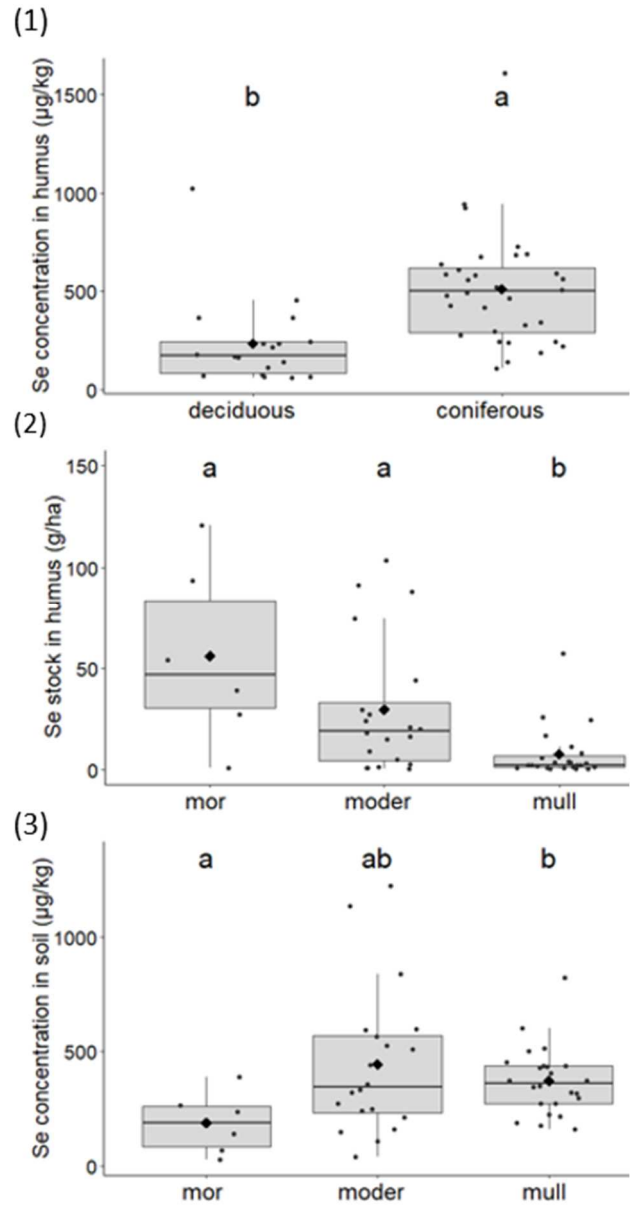


Figure 4 Boxplot representations of: 1) Se concentrations in humus according to forest type; 2) Se stocks in humus according to humus type; 3) Se concentrations in soils according to humus type. Box plots show the first quartile (Q_1), median and third quartile (Q_3) of the data. Black diamond symbols represent average Se concentrations in groups. Lower and upper whiskers extend from the hinges by $Q_1 - 1.5 \times \text{IQR}$ (IQR - interquartile range) and $Q_3 + 1.5 \times \text{IQR}$, respectively. Dots represent corresponding data. The different letters above the boxes correspond to significant differences between the groups according to pairwise Wilcoxon test, with Holm adjustment for p-value.

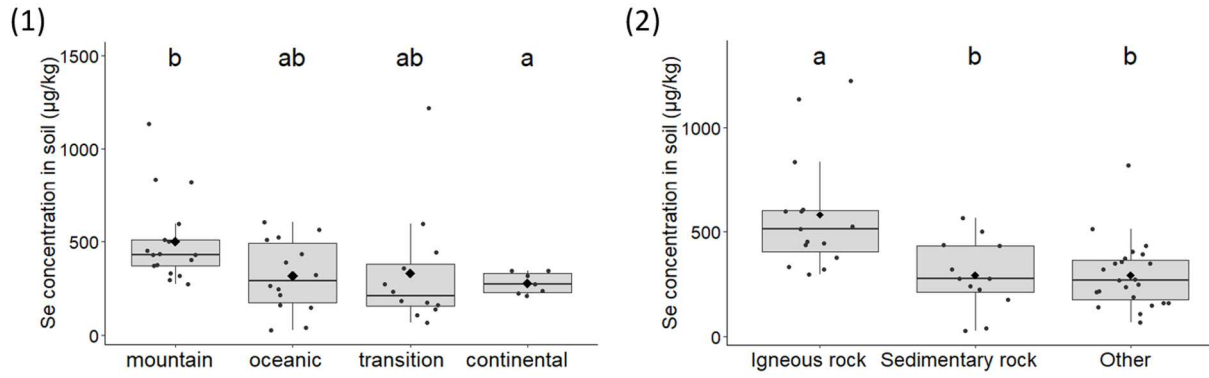


Figure 5 Distribution of Se concentrations in soils according to 1) climate, and 2) rock type. Box plots show the first quartile (Q_1), median and third quartile (Q_3) of the data. Black diamond symbols represent average Se concentrations in groups. Lower and upper whiskers extend from the hinges by $Q_1 - 1.5 \times IQR$ (IQR - interquartile range) and $Q_3 + 1.5 \times IQR$, respectively. Dots represent corresponding data. The different letters above the boxes correspond to significant differences between the groups according to pairwise Wilcoxon test, with Holm adjustment for p-value.

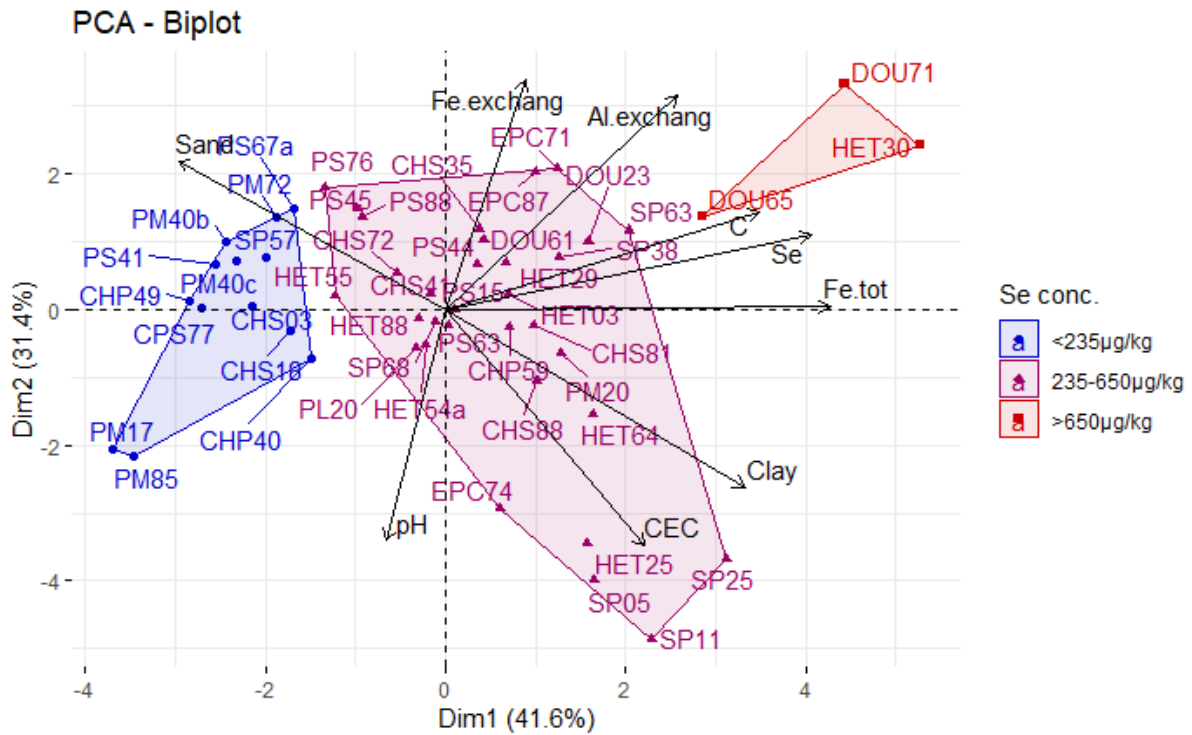


Figure 6 PCA biplot representation of soil composition data on the first two PC axis . Individual soils (48 samples) are colored according to their Se concentration range: : <math><235 \mu\text{g kg}^{-1}</math> - blue, $235\text{-}650 \mu\text{g kg}^{-1}$ - violet, $>650 \mu\text{g kg}^{-1}$ - red. Arrows represent contribution of soil geochemical numerical variables on principal component axes.

Table 1 Statistical analysis for Se concentrations in soil, humus and litterfall. Spearman rank correlation coefficient r_s and Kruskal-Wallis χ^2 values with p-values. Significant correlations are marked in bold.

	Variables	soil		humus		litterfall	
		r_s	p	r_s	p	r_s	p
Numerical variables	Coastal distance	-0.02	0.880	-0.02	0.885	-0.35	< 0.05
	Se conc. in rain ¹	-0.13	0.530	0.23	0.267	0.76	< 0.001
	Se flux in rain ¹	0.34	0.087	0.31	0.123	0.41	< 0.05
	Se flux in litterfall and rain	0.13	0.530	0.21	0.309		
	Se accumulation rate in humus	0.43	< 0.01				
Categorical variable		χ^2	p	χ^2	p	χ^2	p
	Tree species (n=6)	22.58	< 0.001	20.25	< 0.01	7.24	0.203
	Forest type (n=2)	0.17	0.682	15.53	< 0.001	0.15	0.697
	Climate (n=4)	10.74	< 0.05	3.97	0.265	11.36	< 0.01
	Humus type (n=3)	6.02	< 0.05	5.26	0.072		
	Rock type (n=3)	16.34	< 0.001				
	Soil type (n=7)	12.61	0.082				

¹ test performed for 27 sites; n - number of sub-types

Table 2 Contributions on the first three PC axes of i) all variables and ii) reduced number of variables.

Variables	i) contributions of all variables			ii) contribution of reduced number of variables		
	Dim.1	Dim.2	Dim.3	Dim.1	Dim.2	Dim.3
Fe tot	15.806	5.64E-05	0.067	21.585	5.498	0.610
Se	14.990	1.683	0.003	23.708	1.522	1.481
N	13.898	0.744	6.801	-	-	-
Al tot	12.761	3.301	3.463	-	-	-
C _{org}	12.262	5.510	5.761	19.620	0.378	17.124
Clay	8.555	11.535	0.071	-	-	-
Sand	6.962	12.615	9.032	3.853	17.197	50.116
Al exchang.	5.627	10.315	10.237	18.698	5.902	3.600
Silt	3.946	9.073	17.462	-	-	-
CEC	3.637	14.145	8.354	0.282	34.693	0.471
Mn exchang.	0.859	5.798	10.039	-	-	-
pH	0.364	9.701	26.421	5.268	20.562	23.163
Fe exchang.	0.333	15.580	2.288	6.987	14.248	3.435

exchang. – exchangeable