



## Ecotoxicity tests and ecoscores to improve soil management : case of a secondary lead smelter plant

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**ECOTOXICITY TEST AND ECOSCORES TO IMPROVE POLLUTED SOILS MANAGEMENT: CASE OF  
A SECONDARY LEAD SMELTER PLANT.**

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## 1. Introduction

Originally located on the outskirts of cities, numerous industrial sites, sometimes abandoned, are now in urban areas and are therefore likely to have environmental and health risks to surrounding populations [1,2]. Currently, rehabilitation of the sites frequently entails excavation of polluted soils [3]. Excavated soils can thus follow two different ways: landfilling, expensive and energy intensive, or reuse/recycling, integrated to sustainable development. The choice of a specific track mainly depends on total and leachable concentrations of the pollutant in the soil [2]. Among the numerous pollutants observed in urban and peri-urban areas, trace metals are often present in soils [4]; atmosphere emissions by smelters being one of the main anthropogenic source [5, 6]. MTE speciation and compartmentalization in soils can modify their impact on living organisms [5]. Now, numerous publications concluded that these two parameters are strongly influenced by soil organic matter (OM) content, pH and texture [7, 8, 9]. According to Matejczyk et al. [10], chemical weathering of soil minerals favours MTE solubilisation and leachates production. Then, these leachates can pollute surrounding soils and waters. According to the council directive n°1999/31/CE, leaching tests with chemical analysis are therefore currently used for the assessment of environmental hazards of polluted soils. But, landfilling is often inevitable for strongly polluted soils, with high "hazard level" (assessed by leached and total MTE concentrations). Moreover, according to Foucault et al. [1], professionals consider the threshold set as too restrictive and they regret that excavated soils are almost always managed as waste.

In addition to the measure of total and leached MTE concentrations, it appears therefore that knowledge of MTE availability [11] and ecotoxicity may carry useful information [12, 13, 14] to improve environmental risk assessment [10]. Actually, the accurate estimation of metal phytoavailability in polluted soils and solid wastes, using single chemical extraction [15] carry interesting data to perform pertinent risk assessment and remediation efforts [16, 17]. Soil quality integrates both physicochemical and biological characteristics [18]. Moreover, according to Plaza et al. [14] microorganisms play important roles in numerous soil functions. Soils are often polluted with a large variety of compounds leading to possible interactions [19], thus as reviewed by Kim and Owens [20] study of leachates ecotoxicity provides a direct functional characterization of various pollutant mixtures. But, only few studies concern the use of ecotoxicological tests to monitor contamination and bioremediation efficiency of polluted soils [21] and new tests are required by industrial sites managers to assess environmental risks. Among them, microbial bioassays offer quick, cheap and easy ecotoxicity (toxicity and mutagenicity) and bioavailability measurements on bacteria [22, 23]. However, in many cases, microbial bioassays cannot be directly used for the identification and quantification of compounds due to the lack of specificity of the engineered microorganisms [24] and further studies are needed to improve these biotests.

The aim of this study was therefore to assess the ecotoxicity of leachates for landfilling of MTE contaminated soils by various complementary biotests, in addition to usual physicochemical measures. More precisely, the following two scientific objectives were aimed: (1) what is the pertinence of ecotoxicity tests to assess a more realistic human exposition to contaminated soil leachates? (2) What is the influence of soil physicochemical parameters on MTE mobility and leachates ecotoxicity? The originality of this study was to combine the use of new bacterial strains never tested in a context of the remediation of an industrial polluted site and calculation of eco-scores which facilitates the comparisons between different soils.

## 2. Materials and methods

Ten top soil samples were collected in the courtyard of the Chemical Metal Treatments Society (STCM), a secondary lead smelter which currently recycles batteries located in the urban area of Toulouse. This plant was chosen because of its activity and urban location, and many data are already available [1,4,5]. These data allowed defining different areas in terms of environmental and sanitary risks that can vary according to past and present activities. Samples were dried, sieved under 2 mm and treated in triplicate. pH, organic matter and limestone contents, cation exchange capacity and texture, were determined for all soil samples. Pb, As, Cu, Cd, Zn and Sb total concentrations were measured by ICP-OES after mineralization in *aqua regia*.

Normalized leaching test was applied to all soil samples. This procedure consisted of a single extraction with deionised water, using a solid-to-liquid ratio of 1/10. 10 g of soil (granulometry < 4 mm according to the norm) was mixed with 100 mL deionised water during 24h with end-over-end agitation at 5 rpm. After centrifugation at 3,000 g during 15 min, the leachates were filtered with cellulose 0.45 µm

(Millipore®) filters. 10 mL of each leachates were then acidified with  $\text{HNO}_3$  65% prior to analysis by ICP-OES. The other part of leachates was not acidified so as not to disturb microorganisms used for further ecotoxicological tests.

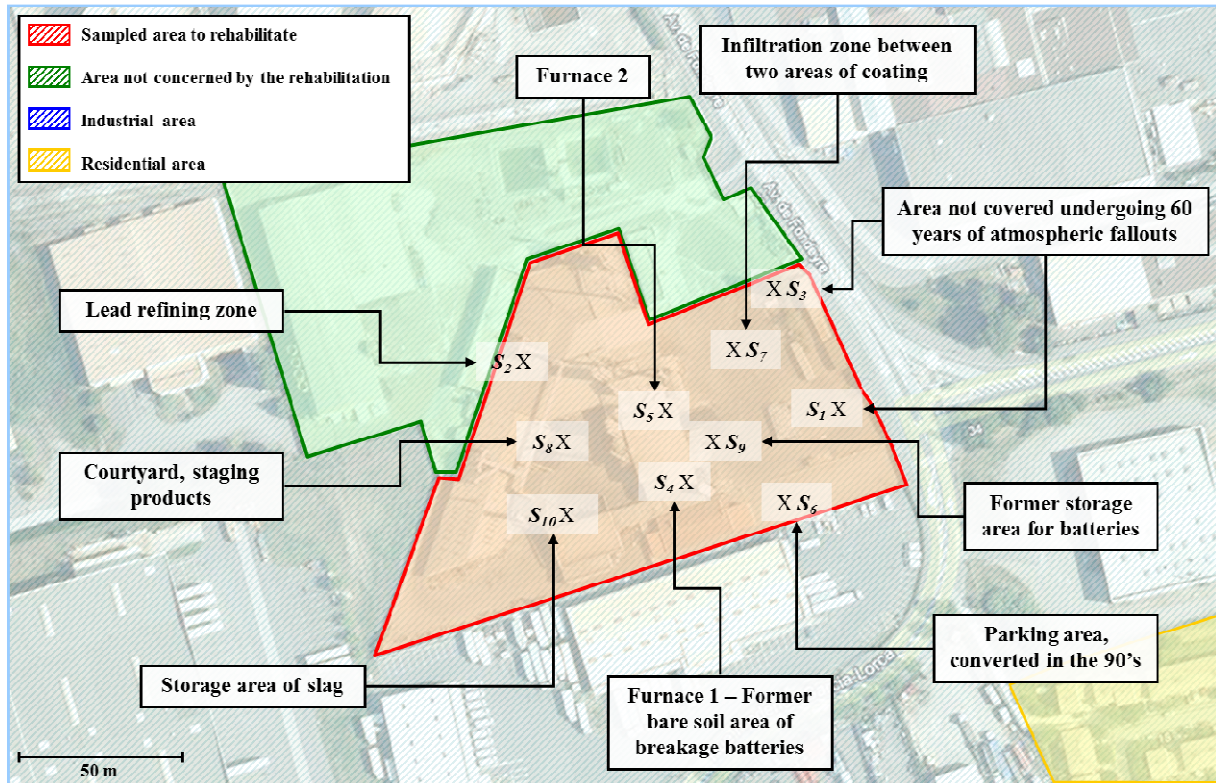


Figure 1 : Situation of the industrial study site, location and characteristics of the 10 sampling points.

Potentially phytoavailable MTE concentrations were estimated by  $\text{CaCl}_2$  extractions according to Uzu et al. [5]. In 25 mL polypropylene centrifugation tubes, 10 mL of  $10^{-2}$  M  $\text{CaCl}_2$  were added to 1.0 g of soil. The liquid to solid ratio of 10 is high enough to avoid samples heterogeneities [25]. After agitation end-over-end during two hours at 5 rpm at 20 °C, samples were then centrifuged during 30 min at 10,000 g. Supernatant was sieved through a  $0.22 \mu\text{m}$  mesh and acidified at 2% with  $\text{HNO}_3$  (15 N, supapur 99.9%). MTE concentrations were finally measured by ICP-OES (IRIS Intrepid II XXDL, analytical errors < 5%).

Acute toxicity tests of leachates were performed on the water flea *Daphnia magna* according to ISO 6341 and with the Microtox® according to the ISO 11348 regulation specifications. Moreover, new bacterial strains which are dedicated to the specific detection of pollutants or pollutant family were used. Thus, a set of five bioluminescent bacteria namely *E. coli* Taflux, *E. coli* Zntlux, *E. coli* Arslux, *E. coli* Coplux and *E. coli* Merlux was used. Monitoring of bioluminescence was recorded and the results were expressed by the logarithm of the induction ratio or the inhibition rate for the inducible strains and the constitutive strain respectively. Decision trees were designed from the learning set of bacterial bioluminescence data using the software “Metalsoft”.

### 3. Results

Physico-chemical characteristics significantly differ in function of sample origin, i.e. their localization on the industrial site in relation with process. pH value varied between 6.9 to 9.2, CEC value varies between 2.6 to  $10.5 \text{ cmol}(+) \text{ kg}^{-1}$  and amounts of soil organic matter and carbonates ( $\text{CaCO}_3$ ) were highly variable: respectively from 0.9 to  $46.7 \text{ g kg}^{-1}$  and from 0 to  $15.0 \text{ g kg}^{-1}$ . MTE concentrations in polluted soil samples were also very heterogeneous: maximum lead concentration is  $42,400 \text{ mgPb.kg}^{-1}$  and other elements are

also present at high levels (up to 2095 mgSb kg<sup>-1</sup>, 288 mgAs kg<sup>-1</sup>, 286 mgCu kg<sup>-1</sup>, 294 mgZn kg<sup>-1</sup> and 80.9 mgCd kg<sup>-1</sup>).

| MTE (mg kg <sup>-1</sup> ) | S <sub>1</sub> | S <sub>2</sub> | S <sub>3</sub> | S <sub>4</sub> | S <sub>5</sub> | S <sub>6</sub> | S <sub>7</sub> | S <sub>8</sub> | S <sub>9</sub> | S <sub>10</sub> |
|----------------------------|----------------|----------------|----------------|----------------|----------------|----------------|----------------|----------------|----------------|-----------------|
| <b>Pb</b>                  | 39800          | 1425           | 42400          | 37250          | 1020           | 297            | 35700          | 1445           | 1750           | 1065            |
| <b>As</b>                  | 288            | 28.7           | 51.5           | 52.5           | 5.8            | 9.3            | 34.3           | 8.65           | 13.3           | 10.2            |
| <b>Cu</b>                  | 286            | 14.7           | 143.5          | 116            | 13.6           | 16.1           | 249            | 19.4           | 59.5           | 60.5            |
| <b>Cd</b>                  | 18.4           | 2.24           | 34.3           | 4.15           | 3.39           | 0.69           | 80.9           | 3.39           | 4.7            | 11.3            |
| <b>Zn</b>                  | 294            | 37.1           | 216            | 218            | 42.8           | 41.9           | 545            | 55             | 116            | 94              |
| <b>Sb</b>                  | 2095           | 53.5           | 1555           | 2175           | 23.5           | 13.1           | 1955           | 15.9           | 44.5           | 9.15            |

Table 1: *Aqua regia* MTE concentrations for the 10 soil samples

Leached MTE amounts in water and corresponding ratios (in comparison with *aqua regia* extraction, considered as “total”) were significantly depending on element nature (Table 2a). The highest extracted concentrations were recorded for lead, antimony and zinc (MTE with high total concentrations): respectively 152.5, 158 and 9 mg kg<sup>-1</sup>, i.e. 8.7 %, 7.3 % and 7.9 %. In comparison, copper (at equivalent content) was significantly less extracted than zinc. Although quantitatively low extracted ( $\leq 5.4$  mgCd kg<sup>-1</sup>), Cd was proportionally one of the most water-soluble element (up to 15.9 % for S<sub>6</sub>). Arsenic was the less extracted MTE with a maximum concentration reached of 2.2 mgAs kg<sup>-1</sup> (7.5 % of the total for S<sub>4</sub>).

CaCl<sub>2</sub> extractions results (Table 2b) showed several contrasted behaviours depending both on chemical element and soil properties. The highest lead quantities extracted by CaCl<sub>2</sub> were observed for S<sub>1</sub>, S<sub>3</sub> and S<sub>7</sub> (up to 178.5 mg kg<sup>-1</sup>). However, for S<sub>4</sub> sample with high total lead concentration, the extracted fraction is low (1.3 mg kg<sup>-1</sup>). Conversely, antimony extracted concentration reached 306.6 mgSb kg<sup>-1</sup>. Other MTE showed a low extractability (in terms of quantity and ratio) whatever the sample, except for S<sub>7</sub> which registered pronounced pools of Cd and Zn associated with high total concentrations. Figures 3-a and 3-b show the fraction of the extracted element in relation to total concentration. Cadmium appeared as the most potentially phytoavailable element. Sb and Zn also represented high extracted fractions respectively for S<sub>4</sub> and S<sub>7</sub>. Moreover, compared to the *aqua regia* fraction, the CaCl<sub>2</sub> fraction remained lower, except for the most potentially phytoavailable Cd element (2–32 %).

| Leaching procedure            | S1    | S2    | S3    | S4    | S5    | S6    | S7    | S8    | S9    | S10   |
|-------------------------------|-------|-------|-------|-------|-------|-------|-------|-------|-------|-------|
| <b>Pb</b> mg kg <sup>-1</sup> | 127,8 | 63,04 | 126,2 | 85,62 | 53,44 | 51,79 | 51,66 | 11,13 | 152,5 | 14,86 |
| % of the total                | 0,32  | 4,42  | 0,30  | 0,23  | 5,24  | 17,44 | 0,14  | 0,77  | 8,72  | 1,40  |
| <b>As</b> mg kg <sup>-1</sup> | 0,29  | 2,17  | 0,28  | 0,33  | nd    | 0,38  | nd    | nd    | 0,54  | nd    |
| % of the total                | 0,10  | 7,55  | 0,55  | 0,62  | -     | 4,13  | -     | -     | 4,07  | -     |
| <b>Cu</b> mg kg <sup>-1</sup> | 1,50  | 0,71  | 1,20  | 0,57  | 1,44  | 1,67  | 0,52  | nd    | 5,04  | 0,87  |
| % of the total                | 0,52  | 4,83  | 0,83  | 0,49  | 10,61 | 10,40 | 0,21  | -     | 8,47  | 1,44  |
| <b>Cd</b> mg kg <sup>-1</sup> | 0,52  | 0,21  | 1,06  | nd    | 0,21  | 0,11  | 5,75  | 0,05  | 0,39  | 0,10  |
| % of the total                | 2,81  | 9,28  | 3,08  | -     | 6,16  | 15,60 | 7,10  | 1,53  | 8,28  | 0,89  |
| <b>Zn</b> mg kg <sup>-1</sup> | 3,71  | 1,90  | 4,19  | 1,26  | 3,32  | 3,86  | 9,04  | 0,41  | 9,13  | 1,00  |
| % of the total                | 1,26  | 5,13  | 1,94  | 0,58  | 7,77  | 9,22  | 1,66  | 0,75  | 7,87  | 1,06  |
| <b>Sb</b> mg kg <sup>-1</sup> | 10,01 | 3,69  | 9,63  | 158,2 | 0,81  | 2,30  | 1,64  | 0,23  | 3,13  | 0,22  |
| % of the total                | 0,48  | 6,90  | 0,62  | 7,28  | 3,43  | 17,53 | 0,08  | 1,47  | 7,03  | 2,40  |

Table 2a: Leached MTE in mg kg<sup>-1</sup> and % of the total concentration

| Phytoavailability assessment  | S1    | S2    | S3    | S4    | S5   | S6    | S7    | S8    | S9   | S10  |
|-------------------------------|-------|-------|-------|-------|------|-------|-------|-------|------|------|
| <b>Pb</b> mg kg <sup>-1</sup> | 178,5 | 0,71  | 162,9 | 1,30  | 0,52 | 0,60  | 89,38 | 14,69 | 0,31 | 3,64 |
| % of the total                | 0,45  | 0,05  | 0,38  | 0,00  | 0,05 | 0,20  | 0,25  | 1,02  | 0,02 | 0,34 |
| <b>As</b> mg kg <sup>-1</sup> | nd    | nd    | nd    | 0,17  | nd   | nd    | nd    | nd    | 0,04 | 0,01 |
| % of the total                | -     | -     | -     | 0,32  | -    | -     | -     | -     | 0,27 | 0,11 |
| <b>Cu</b> mg kg <sup>-1</sup> | 0,43  | 0,202 | 0,37  | 0,38  | 0,19 | 0,18  | 0,32  | 0,21  | 0,29 | 0,64 |
| % of the total                | 0,15  | 1,374 | 0,26  | 0,33  | 1,41 | 1,10  | 0,13  | 1,06  | 0,48 | 1,06 |
| <b>Cd</b> mg kg <sup>-1</sup> | 3,16  | 0,20  | 4,16  | 0,22  | 0,31 | 0,22  | 20,08 | 0,68  | 0,22 | 0,24 |
| % of the total                | 17,22 | 9,02  | 12,14 | 5,21  | 9,22 | 31,91 | 24,82 | 20,00 | 4,68 | 2,15 |
| <b>Zn</b> mg kg <sup>-1</sup> | 2,77  | nd    | 3,51  | 0,06  | 0,20 | nd    | 25,57 | 0,44  | 0,08 | 0,10 |
| % of the total                | 0,94  | -     | 1,62  | 0,03  | 0,48 | -     | 4,69  | 0,79  | 0,07 | 0,11 |
| <b>Sb</b> mg kg <sup>-1</sup> | 1,54  | 0,478 | 1,56  | 306,6 | 0,16 | 0,28  | 0,77  | 0,08  | 0,35 | 0,11 |
| % of the total                | 0,07  | 0,894 | 0,10  | 14,10 | 0,66 | 2,09  | 0,04  | 0,50  | 0,78 | 1,20 |

Table 2b: Extracted MTE with CaCl<sub>2</sub> in mg kg<sup>-1</sup> and % of the total concentration

Ecotoxicity of leachates measured by the inhibition of *Daphnia magna* mobility was highly variable (Table 3). Whatever the sample tested, the inhibition of daphnia mobility increased between 24 h and 48 h, except for S<sub>3</sub> whose inhibition was near 100% after only 24 h. Ecotoxicity was also maximal (i.e. 100%) for S<sub>1</sub>, S<sub>2</sub>, S<sub>7</sub> and S<sub>8</sub> after 48 h; while the lower inhibition was observed for S<sub>5</sub> (15 %). Ecotoxicity was not simply dependant of MTE concentration: (i) the most MTE-enriched leachates were not always the more toxicant; (ii) leachate of S<sub>8</sub> had low MTE concentrations while the inhibition of daphnia mobility was 100 %.

The mean EC<sub>50</sub>-30 min value obtained for zinc sulphate heptahydrate (expressed as Zn<sup>2+</sup>) was 2.38 mg L<sup>-1</sup>, allowing concluding that the invertebrates lot fulfilled the validation specifications. Microtox® test results (Table 3) showed an increase in the number of toxic samples with the contact time: inhibition of bioluminescence was detected in two samples at the beginning of the experiment and for four of them at the end (S<sub>1</sub> and S<sub>7</sub> to 5 min; S<sub>1</sub>, S<sub>4</sub> and S<sub>7</sub> to 15 min; S<sub>1</sub>, S<sub>7</sub>, S<sub>4</sub> and S<sub>3</sub> at 30 min). The measured ecotoxicity also increased over time and was above 90 % for S<sub>1</sub> and S<sub>7</sub> after 30 min of contact. The bacteria were most affected by S<sub>7</sub> with an inhibition of the luminescence of 63% (5 min). Unlike the test on *Daphnia*, S<sub>2</sub>, S<sub>6</sub>, S<sub>8</sub>, S<sub>9</sub> and S<sub>10</sub> showed no toxicity, as S<sub>5</sub> in both bioassays. As described above, S<sub>1</sub>, S<sub>3</sub>, S<sub>4</sub> and S<sub>7</sub> are among the most contaminated leachates ([Pb] > 80 mg kg<sup>-1</sup>; [Sb] > 10 mg kg<sup>-1</sup> (except S<sub>7</sub>)) (Table 2a).

The sensitivity and specificity of inducible bacteria were measured after 60 min in contact with leachates. None of the sample induced the luminescence of Coplux strain. Only two samples, S<sub>1</sub> and S<sub>7</sub>, showed a slight toxicity as demonstrated by the inhibition of luminescence of the constitutive strain pBtaclux (Table 4).

| Ecotoxicity test             |        | Sample         |                |                |                |                |                |                |                |                |                 |
|------------------------------|--------|----------------|----------------|----------------|----------------|----------------|----------------|----------------|----------------|----------------|-----------------|
|                              |        | S <sub>1</sub> | S <sub>2</sub> | S <sub>3</sub> | S <sub>4</sub> | S <sub>5</sub> | S <sub>6</sub> | S <sub>7</sub> | S <sub>8</sub> | S <sub>9</sub> | S <sub>10</sub> |
| <b>D. magna</b> <sup>a</sup> | 24 h   | 90             | 35             | 100            | 40             | 10             | 25             | 80             | 65             | 45             | 5               |
|                              | 48 h   | 100            | 100            | 100            | 65             | 15             | 80             | 100            | 100            | 65             | 70              |
| <b>Microtox</b> <sup>b</sup> | 5 min  | 29             | nt*            | nt             | nt             | nt             | nt             | 63             | nt             | nt             | nt              |
|                              | 15 min | 78.5           | nt             | nt             | 12             | nt             | nt             | 92             | nt             | nt             | nt              |
|                              | 30 min | 93             | nt             | 34             | 34             | nt             | nt             | 96             | nt             | nt             | nt              |

<sup>a</sup>Inhibition of mobility (%); <sup>b</sup>Inhibition of bioluminescence (%).

\*nt: not toxic

Table 3: Results of the *Daphnia magna* and Microtox® tests for the 10 leachates

These samples also induced the luminescence of Zntlux, Arslux and Merlux strains. For S<sub>1</sub> and S<sub>7</sub> the maximum IR was recorded for Arslux (IR = 99.6) and Merlux (IR = 138.1) respectively. S<sub>3</sub> increased moderately the luminescence of Merlux (IR = 12.9) and Zntlux (IR = 4.0) while S<sub>2</sub> induced only Arslux (IR

= 324.9). The analysis with decision trees was then used to determine the elements potentially responsible for ecotoxicity of S<sub>1</sub>, S<sub>2</sub>, S<sub>3</sub> and S<sub>7</sub>. Crosses between results suggested the presence of arsenic in these four leachates (up to 10<sup>-3</sup> M, i.e. more than by chemical analysis), cadmium for S<sub>1</sub>, S<sub>3</sub> and S<sub>7</sub>, biologically at levels lower than those measured chemically (Table 6-b). Analysis of S<sub>7</sub> also showed the presence of copper and mercury (from 10<sup>-4</sup> to 10<sup>-5</sup> M and 10<sup>-9</sup> to 10<sup>-5</sup> M). According to the previous tests, these results also concluded to the ecotoxicity of S<sub>1</sub> and S<sub>7</sub>, and, to a minor extent, the ecotoxicity of S<sub>3</sub> and S<sub>4</sub>.

| Sample          | Bacterial strain    |                     |                     |                       |
|-----------------|---------------------|---------------------|---------------------|-----------------------|
|                 | ZntLux <sup>a</sup> | Arslux <sup>a</sup> | Merlux <sup>a</sup> | pBtaclux <sup>b</sup> |
| S <sub>1</sub>  | 4.8                 | 99.6                | 9.9                 | 6.5                   |
| S <sub>2</sub>  | 0.9                 | 324.9               | 0.8                 | -                     |
| S <sub>3</sub>  | 4.0                 | 1.2                 | 12.9                | -                     |
| S <sub>4</sub>  | 1.4                 | 0.8                 | 0.9                 | -                     |
| S <sub>5</sub>  | 0.7                 | 0.9                 | 1.0                 | -                     |
| S <sub>6</sub>  | 0.6                 | 0.8                 | 0.9                 | -                     |
| S <sub>7</sub>  | 5.4                 | 79.8                | 138.1               | 18.4                  |
| S <sub>8</sub>  | 1,2                 | 0.9                 | 0.9                 | -                     |
| S <sub>9</sub>  | 1,3                 | 2.1                 | 0.8                 | -                     |
| S <sub>10</sub> | 1.2                 | 1.2                 | 0.8                 | -                     |

<sup>a</sup>IF: Induction Factor.

<sup>b</sup>Inhibition rate (%).

Table 4: Ecotoxicity results of bioluminescence emitted by the bacterial strains

#### 4. Discussion

In this study, soil pH were basic or close to neutral conditions and leaching procedure slightly reduced the pH by water addition. Conversely, CaCl<sub>2</sub> is already known to not modify soil pH and give results closer from field reality [15]. Thus, hazard proposed classification of polluted soils differs between water leaching and CaCl<sub>2</sub> procedures. Several studies already showed that MTE extractability is strongly influenced by the nature of the extracting agent which can control element mobility [15, 16]. Moreover, according to Dumat et al. [27] or Ferrari et al. [28], solid-liquid MTE transfers during chemical extractions are complex reactions involving numerous factors that can influence MTE speciation and release. Contact times chosen for chemical extractions were 24h for water and only 2h for CaCl<sub>2</sub> in accordance with the commonly used protocols: these two procedures carry complementary information but the results are not directly comparable.

All MTE were extracted in substantially equal proportions with water (from 0 to 18 %); MTE concentrations in CaCl<sub>2</sub> extracts and corresponding ratios, varied in the range of those reported in the literature [5,34,37], and Cd was the most available element (up to 32 %). Extracted concentrations were strongly correlated to total concentrations. At the reverse side, for all the other elements no relevant correlation was found between total and extracted fractions. In agreement with previous publications [5, 15], these results highlighted the influence of soil properties and MTE nature on its behaviour. Differences observed in function of MTE nature can be explained by different OM or CaCO<sub>3</sub> soil contents, CEC or soil pH. In soils, cadmium is generally easy to dissolve which explain its relatively high extractability [29]. High correlation factors were observed between exchangeable Cu and Zn fractions and soil organic matter amount. These elements were thus less mobile because of their affinity for this soil fraction [11,30]. Concerning lead behaviour, no relationship was found between extracted and total concentrations, and the influence of even one soil parameter was difficult to highlight. Nevertheless, low extraction ratios compared to the most concentrated samples (S<sub>1</sub>, S<sub>3</sub>, S<sub>4</sub> and S<sub>7</sub>) can be explained by stronger bounds on soil phases as mineral fraction [29]. Finally, sorption of metalloids as As and Sb, is mainly controlled by mineral phases [31]. The high Sb amount extracted from S<sub>4</sub> could be explained not only by a higher total concentration but also by the highest CaCO<sub>3</sub> content [32]. Sb could be solubilised under the influence of



soil bio-physico-chemical parameters controlling its sorption [33,34,35]. pH and CEC were already described as influent parameters of element extractability [36], retention and mobility in soils [11]. Thus, according to the origin of soil sampling, differences in soil parameters were observed (Figure 1): in the industrial site, areas not covered or infiltration zones were the most impacted by MTE. Their organic matter content and CEC were also the higher, thus confirming their role in sorption / desorption mechanisms. The choice of the extractant is thereby an important step to be relevant in risk assessment and to avoid an under- or over- estimation of phytotoxicity. Finally, the data obtained by chemical tests are difficult to interpret because of the many parameters interact. The realization of ecotoxicity tests to measure the impact of pollution on ecosystems seems therefore particularly appropriate in this type of study.

The *D. magna* ecotoxicity test was more sensitive to MTE impact than the Microtox® test. But, unlike tests on the different bacterial strains, they do not both provide information on MTE quantification. Ecotoxicity differences were measured for some samples, especially S<sub>2</sub> and S<sub>8</sub>. These differences can be firstly explained by water flea sensitivity. Detection capabilities of the ecotoxicity of the leachate are actually dependent on the test used [38,39] and it has been already shown that *V. fischeri* was generally less sensitive than *D. magna* [10,40]. Instead of these tests, experiments by using bacterial strains allowed to determine and quantify the element which was potentially bioavailable and / or toxic for bacteria [26]. Then, response in ecotoxicity tests was not always directly correlated with total or water-soluble concentrations [41]. These results are in agreement with data previously obtained by Plaza et al. [14] concerning the influence of pH and CEC on MTE behaviour in soils.

Results of this study have shown that this new bioassay enables the screening of samples in terms of environmental risk during remediation process [24]. However, the drawback of the lack of specificity of one strain and the effect of a mixture of MTE (synergistic or antagonistic effects) could be overcome by using a panel of bacterial strains coupled with a predictive model [24]. Due to the lack of specific bacteria for lead, the introduction of other strains induced by lead like *Rastonia Metallidurans* AE 1433 [42] could improve the interpretation of the data.

According to Persoone et al. [43] and Matejczyk et al. [10], the samples were ranked into one of five classes on the basis of the percentage effect (PE) found in *Daphnia* and Microtox® tests. Ranking was based on induction/inhibition rates for bacterial strains. A weight score was calculated for each hazard class to indicate the quantitative importance (weight) of the ecotoxicity in that class. The weight score was expressed as percentage.

(1) Class weight score = (∑ all test scores) / number of tests performed (= 6)

(2) Class weight score (%) = (class weight score) / maximum class weight score × 100

That classification system aimed at the integration of ecotoxicity data obtained in a battery of bioassays as describe by Lors et al. [44]. The classification system is based on two values: a ranking in five acute toxicity classes and a weight score for each toxicity class. The classification of the samples tested in the investigation is reported in Table 5. Samples were classified as slightly and highly toxic in 10 %, toxic in 30 %, and very highly toxic in 50 %. The percentage of class weight class was above 5 % for only S<sub>7</sub> and S<sub>1</sub> (75 % and 62.5 % respectively). These samples were definitely considered as the most hazardous and acutely toxic to the microfauna. The final classification of ecotoxicity risks was S<sub>7</sub> > S<sub>1</sub> > S<sub>3</sub> > S<sub>2</sub> = S<sub>4</sub> = S<sub>9</sub> = S<sub>10</sub> > S<sub>8</sub> > S<sub>5</sub> = S<sub>6</sub>. Although the toxicity of some samples (S<sub>2</sub> and S<sub>8</sub>, for instance) could be different depending on the test used, the ranking based on total concentrations and leachable contents of MTE was almost the same; samples S<sub>1</sub>, S<sub>3</sub>, S<sub>4</sub> and S<sub>7</sub> presenting the greatest risks while the less contaminated areas were generally the less hazardous. However, as our results demonstrated that only a small fraction of total MTE soil concentrations can be solubilised and phytoavailable; ecotoxicity measures complete therefore efficiently standard performed tests for a realistic risks assessment of MTE-contaminated soils accordingly to [10, 14]. Moreover, the use of eco-scores improves the comparisons between bioassays and suggests the use of a restricted battery of tests to perform a cost-effective risk assessment of MTE-contaminated soils.

| Sample                | Score           |                 |               |               |               |                 | Class <sup>a</sup> | MCW <sup>b</sup> | W <sup>c</sup> | PW <sup>d</sup> |
|-----------------------|-----------------|-----------------|---------------|---------------|---------------|-----------------|--------------------|------------------|----------------|-----------------|
|                       | <i>D. magna</i> | <i>Microtox</i> | <i>Zntlux</i> | <i>Arslux</i> | <i>Merlux</i> | <i>pBtaclux</i> |                    |                  |                |                 |
| <b>S<sub>1</sub></b>  | 4               | 3               | 1             | 3             | 2             | 2               | V                  | 4                | 2.5            | 62.5            |
| <b>S<sub>2</sub></b>  | 4               | 0               | 0             | 4             | 0             | 0               | V                  | 4                | 1.3            | 33.3            |
| <b>S<sub>3</sub></b>  | 4               | 1               | 1             | 1             | 2             | 0               | V                  | 4                | 1.5            | 37.5            |
| <b>S<sub>4</sub></b>  | 2               | 1               | 1             | 0             | 0             | 0               | III                | 2                | 0.7            | 33.3            |
| <b>S<sub>5</sub></b>  | 0               | 0               | 0             | 0             | 1             | 0               | II                 | 1                | 0.2            | 16.7            |
| <b>S<sub>6</sub></b>  | 3               | 0               | 0             | 0             | 0             | 0               | IV                 | 3                | 0.5            | 16.7            |
| <b>S<sub>7</sub></b>  | 4               | 3               | 2             | 3             | 4             | 2               | V                  | 4                | 3.0            | 75.0            |
| <b>S<sub>8</sub></b>  | 4               | 0               | 1             | 0             | 0             | 0               | V                  | 4                | 0.8            | 20.8            |
| <b>S<sub>9</sub></b>  | 2               | 0               | 1             | 1             | 0             | 0               | III                | 2                | 0.7            | 33.3            |
| <b>S<sub>10</sub></b> | 2               | 0               | 1             | 1             | 0             | 0               | III                | 2                | 0.7            | 33.3            |

| <sup>a</sup> Class                           | MCW |
|--|-----|
| I no acute toxicity<br>PE < 20 %             | 0   |
| II slight acute toxicity<br>20 % ≤ PE < 50 % | 1   |
| III acute toxicity<br>50 % ≤ PE < 75 %       | 2   |
| IV high acute toxicity<br>75 % ≤ PE < 100 %  | 3   |
| V very high acute toxicity<br>PE ≥ 100 %     | 4   |

<sup>b</sup>MCW: maximum class weight score.

<sup>c</sup>W: class weight score.

<sup>d</sup>PW: class weight score in percent.

Table 5: Ecoscores calculation and hazard classification

## 5. Conclusion

Biotests and eco-scores improve standard tests performed to assess risk on ecosystems induced by polluted soils. In particular, modified bacteria strains sensitive to metals are useful tools highlighting the presence of different MTE and the influence of soil parameters that lead to synergistic or antagonistic effects. Moreover, eco-scores calculation allows an easy and cheap screening of a large number of polluted soil samples and suggests a restricted battery of bioassays to perform a cost-effective risk assessment. Nevertheless, further researches are required to develop a panel of bacterial strains specific of each MTE. Moreover, the chemical characterisation of leachates could provide additional relevant insight with the use of modelling software for determination of MTE speciation and thus to evaluate the predominant parameter influencing soil availability and ecotoxicity. Finally, the procedure developed in the present work could be used in the context of bioremediation techniques like phytoremediation performed on polluted soils, to assess their efficiency and favour their use in a context of sustainable development.

## References

- [1] Y. Foucault, E. Schreck, T. Levêque, P. Pradère, C. Dumat, Toward rational management of excavated soil contaminated by metal trace elements, *Environ. Risk and Health* 11 (2012) 61–66.
- [2] P.A.W. Van Hees, K. Elgh-Dalgren, M. Engwall, T. Von Kronhelm, Re-cycling of remediated soil in Sweden: An environmental advantage? *Resour. Conserv. Recy.* 52, 12 (2008) 1349–1361.
- [3] S. Colombano, A. Saada, V. Guerin, P. Bataillard, G. Bellenfant, S. Beranger, D. Hube, C. Blanc, C. Zornig, I. Girardeau, Quelles techniques pour quels traitements – Analyse coûts-bénéfices, BRGM-RP-58609-FR (2010).
- [4] E. Schreck, Y. Foucault, F. Geret, P. Pradère, C. Dumat, Influence of soil ageing on bioavailability and ecotoxicity of lead carried by process waste metallic ultrafine particles, *Chemosphere* 85, 10 (2011) 1555–1562.
- [5] G. Uzu, S. Sobanska, P. Pradère, C. Dumat, Study of lead phytoavailability for atmospheric industrial micronic and sub-micronic particles in relation with lead speciation, *Environ. Pollut.* 157, 4 (2009) 1178–1185.

- [6] X.S. Luo, S. Yu, X.D. Li, Distribution, availability, and sources of trace metals in different particle size fractions of urban soils in Hong Kong: Implications for assessing the risk to human health, *Environ. Pollut.* 159 (2011) 1317–1326.
- [7] C. Fritsch, P. Giraudoux, M. Coeurdassier, F. Douay, F. Raoul, C. Pruvot, C. Waterlot, A. de Vaulleury, R. Scheifler, Spatial distribution of metals in smelter-impacted soils of woody habitats: Influence of landscape and soil properties, and risk for wildlife, *Chemosphere* 81 (2010) 141–155.
- [8] F. Vega, M. Andrade, E. Covelo, Influence of soil properties on the sorption and retention of cadmium, copper and lead, separately and together, by 20 soil horizons: Comparison of linear regression and tree regression analyses, *J. Hazard. Mater.* 174, 1–3 (2010) 522–533.
- [9] M. Sammut, Y. Noack, J. Rose, J. Hazemann, O. Proux, M. Depoux Ziebel, E. Fiani, Speciation of Cd and Pb in dust emitted from sinter plant, *Chemosphere* 78, 4 (2010) 445–450.
- [10] M. Matejczyk, G.A. Plaza, G. Nalęcz-Jawecki, K. Ulfig, A. Markowska-Szczupak, Estimation of the environmental risk posed by landfills using chemical, microbiological and ecotoxicological testing of leachates, *Chemosphere* 82 (2011) 1017–1023.
- [11] [M.A. Kashem, B.R. Singh, T. Kondo, S.M. Imamul Huq, S. Kawai, Comparison of extractability of Cd, Cu, Pb and Zn with sequential extraction in contaminated and non-contaminated soils, *Int. J. Environ. Sci. Technol.* 4, 2 (2007) 169–176.
- [12] CEN 14735, Characterization of Waste. Preparation of Waste Samples for Ecotoxicity Tests, European Committee for Standardization, Brussels (2006) 47pp.
- [13] I. Buj, J. Torras, M. Rovira, J. de Pablo, Leaching behaviour of magnesium phosphate cements containing high quantities of heavy metals, *J. Hazard. Mater.* 175, 1–3 (2010) 789–794.
- [14] G.A. Plaza, G. Nalęcz-Jawecki, O. Pinyakong, P. Illmer, R. Margesin, Ecotoxicological and microbiological characterization of soils from heavy-metal and hydrocarbon-contaminated sites, *Environ. Monit. Assess.* 163 (2010) 477–488.
- [15] E. Meers, R. Samson, F.M.G. Tack, A. Ruttens, M. Vandegheuchte, J. Vangronsveld, M.G Verloo, Phytoavailability assessment of heavy metals in soils by single extractions and accumulation by *Phaseolus vulgaris*, *Environ. Exp. Bot.* 60, 3 (2007) 385–396.
- [16] N.W. Menzies, M.J. Donn, P.M. Kopittke, Evaluation of extractants for estimation of the phytoavailable trace metals in soils, *Environ. Pollut.* 145, 1 (2007) 121–130.
- [17] M. Arshad, J. Silvestre, E. Pinelli, J. Kallerhoff, M. Kaemmerer, A. Tarigo, M. Shahid, M. Guisresse, P. Pradère, C. Dumat, A field study of lead phytoextraction by various scented *Pelargonium* cultivars, *Chemosphere* 71, 11 (2008) 2187–2192.
- [18] J. Römbke, A.M. Breure, C. Mulder, M. Rutgers, Legislation and ecological quality assessment of soil: implementation of ecological indication systems in Europe, *Ecotox. Environ. Safe* 62 (2005) 201–210.
- [19] S.A. Mansour, M.F. Gad, Risk assessment of pesticides and heavy metals contaminants in vegetables: A novel bioassay method using *Daphnia magna* Straus, *Food Chem. Toxicol.* 48, 1 (2010) 377–389.
- [20] K.R. Kim, G. Owens, Potential for enhanced phytoremediation of landfills using biosolids - a review., *J. Environ. Manage.* 91 (2010) 791–797.
- [21] L. Leitgib, J. Kalman, K. Gruiz, K., Comparison of bioassays by testing whole soil and their water extract from contaminated sites, *Chemosphere* 66, 3 (2007) 428–434.
- [22] M.J. Durand, G. Thouand, T. Dancheva-Ivanova, P. Vachon, M. DuBow, Specific detection of organotin compounds with a recombinant luminescent bacteria, *Chemosphere* 52, 1 (2003) 103–111.
- [23] T. Elad, E. Benovich, S. Magrisso, S. Belkin, Toxicant identification by a luminescent bacterial bioreporter panel: application of pattern classification algorithms, *Environ. Sci. Technol.* 42, 22 (2008) 8486–8491.
- [24] S. Jouanneau, M.J. Durand, P. Courcoux, T. Blusseau, G. Thouand, Improvement of the identification of four heavy metals in environmental samples by using predictive decision tree models coupled with a set of five bioluminescent Bacteria, *Environ. Sci. Technol.* 45, 7 (2011) 2925–2931.
- [25] V.J.G. Houba, T.M. Lexmond, I. Novozamsky, J.J. Van Der Lee, State of the art and future developments in soil analysis for bioavailability assessment, *Sci. Total Environ.* 178, 1–3 (1996) 21–28.
- [26] T. Charrier, M.J. Durand, M. Affi, S. Jouanneau, H. Gezekel, G. Thouand, Bacterial Bioluminescent Biosensor Characterisation for On-line Monitoring of Heavy Metals Pollutions in Waste Water Treatment Plant Effluents. In *Biosensors*; Serra, P.A., Ed.; INTECH. (2010) pp 179–206.

- [27] C. Dumat, K. Quenea, A. Bermond, S. Toinen, M. Benedetti, Study of the trace metal ion influence on the turnover of soil organic matter in cultivated contaminated soils., *Environ. Pollut.* 142, 3 (2006) 521–529.
- [28] B.J.D. Ferrari, J.F. Masfaraud, A. Maul, J.F. Féraud, Predicting uncertainty in the ecotoxicological assessment of solid waste leachates, *Environ. Sci. Technol.* 40, 22 (2006) 7012–7017.
- [29] L. Beesley, E. Moreno-Jiménez Clemente, R.N. Lepp, N. Dickinson, Mobility of arsenic, cadmium and zinc in a multi-element contaminated soil profile assessed by in-situ soil pore water sampling, column leaching and sequential extraction, *Environ. Pollution* 158, 1 (2010) 155–160.
- [30] P.F.A.M. Romkens, H.Y. Guo, C.L. Chu, T.S. Liu, C.F. Chiang, G.F. Koopmans, Prediction of cadmium uptake by brown rice and derivation of soil-plant transfer models to improve soil protection guidelines, *Environ. Pollution* 157, 8–9 (2009) 2435–2444.
- [31] S.C. Wilson, P.V. Lockwood, P.M. Ashley, M. Tighe, The chemistry and behavior of antimony in the soil environment with comparison to arsenic: a critical review, *Environ. Pollution* 158, 5 (2010) 1169–1181.
- [32] S. Denys, K. Tack, J. Caboche, P. Delalain, Bioaccessibility, solid phase distribution, and speciation of Sb in soils and digestive fluids. *Chemosphere* 74 (2008) 711–716.
- [33] B. Collin, E. Doelsch, Impact of high natural soilborne heavy metal concentration on the mobility and phytoavailability of these elements for sugarcane. *Geoderma* 159, 3–4 (2010) 452–458.
- [34] S.M. Rodrigues, B. Henriques, E. Ferreira da Silva, M.E. Pereira, A.C. Duarte, J.E. Groenenberg, P.F.A.M. Römken, Evaluation of an approach for the characterization of reactive and available pools of 20 potentially toxic elements in soils: Part II – Solid-solution partition relationships and ion activity in soil solutions, *Chemosphere* 81, 11 (2010) 1560–1570.
- [35] X.S. Luo, S. Yu, X.D. Li, The mobility, bioavailability, and human bioaccessibility of trace metals in urban soils of Hong Kong. *Appl. Geochem.* 27, 5 (2012) 995–1004.
- [36] J.S. Rieuwerts, M.R. Ashmore, M.E. Farago, I. Thornton, The influence of soil characteristics on the extractability of Cd, Pb and Zn in upland and moorland soils, *Sci. Tot. Environ.* 266, 2–3 (2006) 864–875.
- [37] [61] A. Lopareva-Pohu, A. Verdin, G. Garçon, A. Lounès-Hadj Sahraoui, B. Pourrut, D. Debiane, C. Waterlot, F. Laruelle, G. Bidar, F. Douay, P. Shirali, Influence of fly ash aided phytostabilisation of Pb, Cd and Zn highly contaminated soils on *Lolium perenne* and *Trifolium repens* metal transfer and physiological stress. *Environ. Pollut.* 159, 6 (2011) 1721–1729.
- [38] M. Isidori, M. Lavorgna, A. Nardelli, A. Parrella, Toxicity identification evaluation of leachates from municipal waste landfills: a multispecies approach, *Chemosphere* 52, 1 (2003) 85–94.
- [39] L. Manusadzianas, K. Sadauskas, R. Vitkus, Comparative study of indices used in toxicity evaluation of effluents, *Desalination* 250, 1 (2010) 383–389.
- [40] R. Boluda, L. Roca-Pérez, L. Marimón, Soil plate bioassay: An effective method to determine ecotoxicological risks, *Chemosphere* 84, 1 (2011) 1–8.
- [41] [E.E.V. Chapman, G. Dave, J.D. Murimboh, Ecotoxicological risk assessment of undisturbed metal contaminated soil at two remote lighthouse sites, *Ecotox. Environ. Safe.* 73 (2010) 961–969.
- [42] P. Corbisier, E. Thiry, L. Diels, Bacterial biosensors for toxicity assessment of solid waste, *Environ. Toxic. Water* 11 (1996) 171–177.
- [43] G. Persoone, B. Marsalek, I. Blinova, A. Torokne, D. Zarina, L. Manusadzianas, G. Nalęcz-Jawecki, L. Tofan, N. Stepanova, L. Tothova, B. Kolar, A practical and user-friendly toxicity classification system with microbiotests for natural waters and wastewaters, *Environ. Toxicol.* 18, 6 (2003) 395–402.
- [44] C. Lors, J.F. Ponge, M. Martínez Aldaya, D. Damidot, Comparison of solid and liquid-phase bioassays using ecoscores to assess contaminated soils. *Environ. Pollut.* 159 (2011) 2974–2981.