TOWARD AN IMPROVEMENT OF ENVIRONMENTAL ASSESSMENT OF ALTERNATIVE MATERIALS IN ROAD CONSTRUCTION

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ABSTRACT

A back analysis of the monitoring of road structures made with alternative materials was carried out at national scale (France). It underlines the need to harmonize the practices of study around comparable assessment methods and criteria. The assessment of the environmental acceptability of alternative material use in road construction must follow, by adapting it, the eco-compatibility approach used for study and management of waste and by-products: characterization of *Source, Transport and Impact* terms. The specificities of the road use scenario lead to consider the road soil as the immediate target to be possibly hit by alternative material percolates. On-site diagnosis and laboratory simulations by means of column underline the *capacity* of road soils to fix heavy metals. The *stability* of fixing is also assessed, by means of fractioning. The *acceptability* of the contaminated *state* of road soils is assessed thanks to comparison with reference values. In order to clarify the *effect* of the road made of alternative materials on road soils and to allow inter-material and inter-site comparisons, a specific method of assessment by means of indicators is developed; it is applied to field data and discussed.

1. INTRODUCTION

The growing demand for valorisation of alternative material in road construction calls for the developments of tools for their characterisation and for the prediction of their field behaviour. From the environmental point of view, the fair assessment of these materials calls for a clarification of some basic concepts in order to focus development efforts on realistic domains. In this view research can take advantage of lessons from past experiments, and can contribute to the development of tools for decision making. The successive steps of a progression toward the development of specific indicators for the assessment of the *effects* of roads made of alternative materials on road soils, are presented below. This progression passes through the evaluation of the *capacity* of road soils to fix contaminants, through the assessment of the *stability* of fixing, and through the assessment of the *acceptability* of such a contamination.

2. CONTEXT OF ALTERNATIVE MATÉRIAL USE

The aims assigned by the European Union in the sixth Community programme of action for the environment regarding natural resource and waste management are to take care that the consumption of renewable and non-renewable resources does not exceed what the environment is able to support. This can be achieved in particular thanks to an increased effectiveness in the use of resources and with the reduction of waste production. The aim is to reduce the production of waste of 20% by 2010 and 50% by 2050 [1]. Among the recommended actions appears the definition of rules of good practice, which implies to better understand, and to be able to predict, the mechanisms of evolution of waste under specific use conditions. This point is particularly important in view of the use of alternative materials (wastes and by-products) in road construction.

Alternative material, contrary to traditional natural materials is seldom inert - at least in the short term; implemented in a given road layer (base course, sub-base…) it can show different performances depending upon variations of some local external factors. Thus, with time, appeared many questions relating to the forecast of the actual mechanical performances and the potential effects on the environment of these materials, from the short to the long term.

Traditional methods of material assessment did not make it possible to bring satisfactory answers to these questions [2]. Since the nineties this gave rise to various demonstration programs and studies aiming at clarifying the technical and environmental relevance of this way of valorisation. In France, some of these studies were identified in the OFRIR database (http://ofrir.lcpc.fr). This database available since 2003 gathers and diffuses classified and validated information about 16 alternative materials in order to help recycling decision [3]. However very little data about road structures made of alternative materials was gathered. For that reason - although it is a crucial point for the assessment of the behaviour of materials in road use scenarios - the analysis of the experience feedback of alternative materials in road construction was not carried out until now at a national scale.

3. BACK ANALYSIS OF ALTERNATIVE MATERIALS IN ROAD STRUCTURES

3.1. Aims and approach

The aim of the study (called CAREX), launched by LCPC at the national environment agency (ADEME) request in 2004-2005, was to carry out this first back analysis, with two objectives. One was to extract general rules of behaviour for different alternative materials. Another one was to analyze practices of study related to testing at road scale, in order to assess their efficiency and possibly improve future experiments. An investigation was led at national scale in order to identify all available documents relating to a mechanical and/or environmental monitoring of road structures. Any type of structure was considered, from various types of functional infrastructures subjected to a road traffic (roads, car-parks), to trial areas, the latter being defined as structures located apart from traffic and exclusively dedicated to study.

3.2. Some outcomes of the back analysis

The investigation enabled to find out 17 cases of study (of which 2 only didn't provide environmental data), varied in terms of materials, types of structures (motorway sections - MS, trunk road sections - TRS, departmental road sections - DRS, streets, private ways, and trial areas not submitted to traffic), application layers, types of study (documentation of the construction phase, monitoring of structures, diagnosis study) and location (essentially Paris region, north, east and west of France). An overview of the whole cases of study is provided in Table 1. Despite this diversity, a great part of the cases of study related to Municipal Solid Waste Incinerator (MSWI) bottom ash, in relation to the concerns particular to this material in France since the nineties.

As an answer to the second objective of the CAREX study, the back analysis opens prospects regarding the improvement of steering for future experimentations in order to strengthen their scientific benefit and their realism. But the main outcome underlines the need to harmonize practices of study around comparable assessment methods and criteria, as well as problems of scaling between the laboratory and the on-site behaviour. The diversity of methods of study makes difficult inter-site and inter-material comparisons.

Table 1 – Overview of CAREX cases of study

Alternative material	Structure	Layer	Location
MSWI bottom ash	MS	Embankment + Capping layer	East
Chromium tailings	MS	Embankment + Capping layer	North
MSWI bottom ash	TRS	Embankment	East
MSWI bottom ash	TRS	Embankment + Capping layer	East
MSWI bottom ash	DRS	Capping layer	Centre
MSWI bottom ash	DRS	Capping layer	Paris region
Steel slag	DRS	Capping layer + Subbase	Rhône-Alpes
MSWI bottom ash	street	Capping layer	Paris region
MSWI bottom ash	street	Capping layer	Rhône-Alpes
Bound MSWI bottom ash	street	Subbase + Base	South-west
MSWI bottom ash	private way	Capping layer + Subbase	West
MSWI and Industrial WI bottom ash	private way	Subbase	West
MSWI bottom ash	private way	Subbase + Base	South-west
MSWI bottom ash	trial area	Subbase	Paris region
Coal fly ash	trial area	Capping layer to Base	Rhône-Alpes
Zinc and lead flush slag	trial area	Base	North
Treated MSWI fly ash	private way	Base	East

4. THÉORETICAL SCHEMA FOR ENVIRONMENTAL ASSESSMENT

In order to assess the environmental *acceptability* of alternative material use for road construction, it seems rational to follow the approach used in the general framework of wastes and by-products management. Such principles must however be applied to the road scenario with consideration of some important specificities.

4.1. The eco-compatibility approach

The eco-compatibility approach developed by ADEME [4], applies to the environmental assessment of scenarios of dumping and valorisation of wastes. It is dedicated to mineral wastes in scenarios in which the main pollutant vector is water. The eco-compatible state must be obtained for all kinds of scenario. It is defined as « a situation where pollutant fluxes released by wastes – when the latter are placed in a certain physical, hydro-geological, physico-chemical and biological context – are compatible with pollutant fluxes that are acceptable for receiving environments of the concerned site ». The general principle of the method is based on the successive study of three terms: the *Source*; the *Transport*; the *Impact*. The *Source* term is the structure that contains wastes; under the effect of rainfall infiltration, it is expected to release a certain flux of pollutant (F_S). The *Transport* term develops into the soil of transfer; the latter is expected to receive F_S and to transform it into F_T (the flux that actually reaches natural environments). Thanks to physico-chemical reactions with the soil [5], F_T is expected to be lower than F_S. The *Impact term* assessment lies in the confrontation of F_T with the fluxes that are acceptable for aquatic environments (F_{CA}) and those that are acceptable for terrestrial environments

(F_{CB}): wastes in a given scenario are acceptable if, and only if, $F_T \leq F_{CA}$ and/or $F_T \leq F_{CB}$. Impacts on aquatic and terrestrial environments can be varied; the definition of impacts implies the selection of certain targets. Regarding terrestrial environments, the natural soil, as support of life for terrestrial organisms, can be considered as a target.

The development of the European Landfill Directive [6,7] which aims at developing operational acceptance criteria for wastes, is consistent with the eco-compatibility methodology (*Source, Transport, Impact*). In the Landfill Directive approach, only the impacts on surface water and groundwater is considered. For the latter, in practice, a decision must be made concerning the setting of points of compliance (POC), i.e. the downstream points where the groundwater quality criteria must be fulfilled [8]. No similar approach is operational regarding alternative material use in road construction today.

4.2. Specificities of the road scenario

Assessment of pollutant releases from a road made of alternative materials (*outputs*) and of their possible effects on targets, must be done with consideration of possible pollutant *inputs* in the system. Indeed, in addition to the possible emission of pollutants during the road construction phase, bound to the normal service of the road, three possible origins of pollution are generally considered [9]. These are the chronicle pollution due to traffic and road equipments wearing; the seasonal pollution due to spring operations of chemical weeding or winter operation of de-icing; and the accidental pollution due to dumping of hazardous products. Some pollutants may infiltrate into the road surface, transfer through the road body and continue their way toward downstream natural targets. The alternative material applied in the road structure must not be incriminated for such emissions; an assessment methodology must allow discrimination.

Landfills (waste containment centres) are made of racks which generally are dug in the ground and sealed off by a geo-membrane, so that in case of release, the groundwater is the immediate target to be concerned. On the contrary, in road engineering, for mechanical purpose, structures are designed in order to prevent any water saturation: a sufficient unsaturated zone is kept below the structure; anti-capillary rise devices are used; if necessary a draining system is applied. The proximity of the water table with road layers happens in very seldom contexts of lowlands where no solution of drainage can apply.

Under the effect of rainfall infiltration into the road body, alternative materials can release contaminant toward road soil at first, and then possibly toward groundwater. Thus, in the road use scenario, groundwater can not be considered as the immediate target; the first risk to consider is the contamination of the soil underlying the structure.

A last specificity of the road use scenario is related to two important characteristics of road soils toward interactions with heavy metals. Most alternative materials are mineral materials (demolition residues, extraction industry residues, thermal processes residues), and the capacity of natural soils to fix heavy metals is essentially linked to their organic matter and clayey minerals contents [10, 11]. Now, before the construction of the first layer of the road body, in order to prevent deleterious effects, road engineering requires to remove the superficial horizons rich in organic material and clay (stripping). The natural soil is scraped down to 30 cm at least, then compacted. Figure 1 illustrates the main features of the road use scenario in relation to infiltration and percolation.

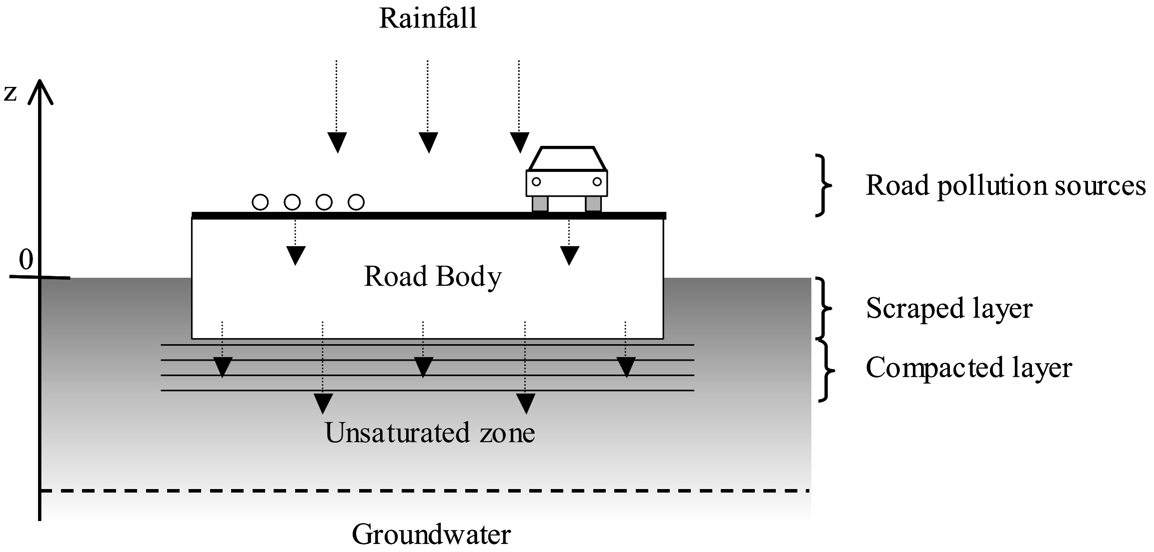

Figure 1 – Infiltration and percolation through the road body

Poor in clayey minerals and in organic matter, road soils may therefore be less efficient than natural ones to fix heavy metals. Compared to natural soils, compaction may also reduce the velocity of leachate infiltration. Due to the important intrinsic differences between natural soils (well documented) and road soils (not documented) regarding the potential interaction with alternative material leachates, a field diagnosis was necessary in order to gauge the actual extent of impacts and to identify the main interaction phenomena. Hence, diagnosis on old sites of use were planned. And then laboratory simulations on large columns reconstituting the characteristics of road soils were used in order to confirm and interpret field observations.

5. ON-SITE DIAGNOSIS

5.1. Long-term diagnosis studies

In order to assess the impact on road soils of structures made of alternative materials, some field investigations were planned in France in 1998 to identify sites of interest [12]. To make the most of optimised conditions of marking of pollutants transfer from alternative material to road soil, some criteria were defined for the selection of sites. The road structure had to be submitted to a significant traffic in order to integrate some possible fatigue consequences like cracking. The layer made of alternative material had to be in direct contact with the road soil in order to avoid leachate interaction with some intermediate layers. The site had to be the oldest as possible in order to allow the longer time of contact as possible. Due to its high leaching potential the alternative material considered in this survey was MSWI bottom ash [13]. The search for old sites, led to two roads made of MSWI residues, i.e. produced at a time when MSWI bottom ash and MSWI fly ash were not separated (before the enforcement of a 1991 ministerial order [14]). Their pollutant potential was higher than the one of the single MSWI bottom ash allowed since 1994 [15].

Two sites located within contrasted pedological settings were selected. Site A was the private way of an incineration plant. It was built in 1976 (22-year old at the moment of the diagnosis). The pavement was composed of a 20 cm thick MSWI residue sub-base; a 8 cm thick base of unbound graded aggregate and a 4 cm thick asphalt layer. The road soil was entirely sandy. Site B was a urban arterial built in 1978 (20-year old). The pavement was made of a 25 cm thick MSWI residue sub-base; a 15 cm thick base of unbound graded aggregate; and a 15 cm thick asphalt surface course. The road soil was sandy over the upper 40 cm and silty below [12]. Some heavy metal contents in Site B road soil are presented in Table 2.

Vertical profiles of some heavy metal contents show decreasing evolutions from the contact with the alternative material layer toward the deeper soil levels (e.g. cadmium and copper). In the soils levels where the clay content is higher (e.g. in Site B it is comprised between 12.5 % and 18.2 % in the silty levels), even if they are deeper than the sandy levels, the road soil shows higher contents for some metals (chromium, nickel, zinc).

Table 2 – Assessment of soil contents for a site (B)

Soil		Soil content[a] (mg/kg) / ILV [16] (mg/kg)					
Levels	Texture	Cd	Cr	Cu	Ni	Pb	Zn
-60/-70 cm	sandy	0.3 / 8	43^+ / 233	16.3^+ / 107	8.1 / 94	32 / 363	52^+ / 269
-70/-80 cm	sandy	0.06 / 8	38^+ / 228	9.6^+ / 105	8.7 / 90	26 / 360	50^+ / 255
-80/-100 cm	silty	0.04 / 9	95^+ / 292	8.8 / 138	29.6^+ / 140	26 / 425	87^+ / 447
-100/-120 cm	silty	0.01^- / 10	132^+ / 328	9.5 / 157	37.9^+ / 169	26 / 463	108^+ / 555

a = + : Soil content above the 9th decile of ordinary soils in France [11] ; a = - : Soil content below the 1st decile of ordinary soils in France [11].

From the point of view of the sustainable management of the environment, the *acceptability* of accumulation of heavy metals in soils must be assessed. This can be done thanks to comparison to reference values.

5.2. Comparison between results and reference values

Comparison between soil contents and acceptance limit values compatible with functional soil properties for flora, fauna and human use, constitutes such a method. Such threshold values are not available in France today, but in the Netherlands, some have been set in the Soil Protection Act (SPA) for some metals; these Intervention Limit Values (ILV) are calculated on the basis of organic material content and clay fraction of soils [16]. A second approach can consist in comparing road soil contents with contents recorded elsewhere, in other soils with the same texture. With such an approach, the interest of the comparison depends on the statistical representativeness of the reference value. In France, such statistics have been gathered for soils of different natures by the INRA research institute [11]. In Table 2, the ILV calculated for each metal at each soil level is provided as denominator, and the exponent (a) indicates whether the soil content belongs to the 10% more loaded soils in France (a = +) or to the 10% less loaded ones (a = -).

Results observed in the two old sites show that zinc and chromium often exceed the 9th decile of usual contents of ordinary soils of the same texture in France [11]. As a whole, contents above the 9th decile of similar ordinary soils (a = +) represent only a quarter of all the measured contents in Cd, Cr, Cu, Ni, Pb and Zn. Moreover, these contents are all significantly below the pollution threshold values calculated from the methodology used in the Dutch SPA: they represent from 5 to 82% (24% in average) of the corresponding ILV. The rate of 82% is observed in a single case: for zinc in the 5 cm thick soil layer immediately below the alternative material layer in Site A.

These results provide a consistent and rather positive picture on the medium term fate (20 years) of heavy metals in road soils from initial conditions not very favourable (fraction of MSWI fly ash in the material, low organic and clay contents in road soils). From the comparison with regulatory and statistical reference values, all measured contents appear as fully acceptable.

6. POLLUTANT RETENTION STUDY THROUGH LABORATORY SIMULATIONS

Despite on-site diagnosis provide acceptable results regarding road soil contents after two decades, in terms of sustainable management of the environment, the dynamic of fixing and its *stability* at longer term must be assessed. Laboratory simulations on road soils reconstituted in column were used for that purpose.

6.1. Reconstitution of road soils in columns

On the basis of field observations, it was important to develop the understanding of the ability of road soil to act as a long term pollution barrier toward the migration of heavy metals transported by percolates. For that purpose, a research aiming at studying the evolution of a leachate during its infiltration into a road soil, and at analysing the stability of fixing into the soil matrix, was launched in 2001 [17]. Road soils – different in terms of physical and chemical properties - were sampled on road works and compacted on a thickness of 50 cm according to standard Proctor references (90% of Standard Proctor Optimum to allow enough porosity and permeability) into plexiglas columns (ϕ 24 cm). The latter were fed with a leachate produced from a fresh production MSWI bottom ash sample. Discontinuous injections (1 litre) were used in order to maintain unsaturated conditions representative of usual infiltration conditions into road bodies. In order to follow the evolution of pH and heavy metal concentration, the percolating solution was sampled at intermediate depths (-15 cm and -25 cm) using soil solution samplers, and at the outlet of the column (-50 cm). The design of columns is presented in Figure 2.

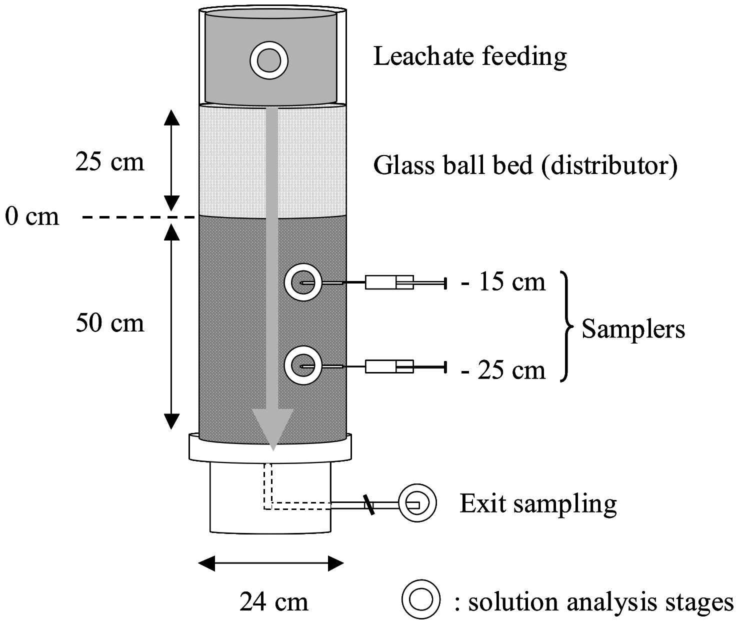

Figure 2 – Column design for laboratory simulations

6.2. Evolution of the leachate during infiltration

Depending on the permeability of the compacted soil (which determined the rhythm of injection of MSWI bottom ash leachate), at the end of the “contamination” process, the different columns reached final liquid/solid (L/S) ratios of 0.34 L.kg^{-1}, 2.75 L.kg^{-1} and 5.50 L.kg^{-1}. During the whole ”contamination” process monitoring, no heavy metal was detected in any solution sample but a pH evolution was noticed. The pH of the MSWI bottom ash solution incorporated at the top of columns was 12.5 (controlled by Portlandite - $Ca(OH)_2$ – dissolution) while the pH of both road soil solutions before "contamination" was around 8. In the first column, the monitoring at -15 cm; -25 cm and -50 cm (outlet) shows that the solution pH remains unchanged during the whole "contamination" process. The monitoring of the second column shows no pH change at the outlet of the column during the whole experiment. However, at -15 cm, when L/S reaches the value of 1.5 $L.kg^{-1}$, the pH climbs to 10.5. Then it remains stable until the end of the experiment. At -25 cm, a continuous pH increase is detected from L/S = 1.5 $L.kg^{-1}$ until the end of monitoring: pH 10 at L/S = 2.75 $L.kg^{-1}$. It seems that if the experiment would have last more time, the pH at this depth would have reached also the value of 10.5. The third column shows the same phenomena, however, in this case the experiment is long enough to allow the pH at -25 cm to reach the equilibrium value of 10.5. In this case too, deeper, the pH remains equal to its initial value (≈ 8).

6.3. Mass transfer toward soil and comparison with field percolation

After completion of the "contamination" process, soils samples were obtained by cutting in the column 1 cm thick slices, in order first to determine the vertical distribution of heavy metals content in the soil. For the different columns, final vertical profiles of Cr, Cu, Pb and Zn show no content increase below -5 cm; the increase is confined in the first centimetres of soil. Compared to the initial state the increase is simply higher when the final L/S ratio is higher. As an example, for Pb, the first centimetre of soil contains 78 $mg.kg^{-1}$ in the first column; 86 $mg.kg^{-1}$ in the second; and 400 $mg.kg^{-1}$ in the third. These results are consistent with those observed on-site [12]; they underline the *capacity* of road soils to fix heavy metals within the very first centimetres.

Under real conditions, rainwater infiltrates through pavements according to a law which main parameters are the saturated hydraulic conductivity (Ks) of the surface course on one hand, and the intensity ($mm.s^{-1}$) and duration of rainfall on the other hand [18]. Thus, for a given rainfall context defined by its annual rainfall and its usual rainfall intensity, it is possible to estimate, for a given pavement permeability, an annual amount of infiltration. A theoretical estimation can be done for a recent surface course with permeability included between 10^{-8} and 10^{-7} $m.s^{-1}$, in the rainfall context of Nantes (west of France). Following abacus provided by van Ganse [18], this leads to an annual infiltration of around 175 mm. Lower road layers being supposed to be more permeable than surface course; any alternative material used below can be submitted to this annual flow. Considering a usual MSWI bottom ash application with a dry density (ρ_d) of 1.7 (average dry density measured on-site [12]) and 40-cm thick (e.g. sub-base), that is 680 kg per square meter of pavement, the theoretical annual L/S ratio of the MSWI bottom ash would be 0.25 $L.kg^{-1}$ [19]. This average ratio can be used to provide a rough estimate of the on-site temporal prospect represented by the L/S ratios reached by column simulations, to 1 year, 10 years and 20 years respectively. This last case corresponds to the age of studied road sites (A and B) at the moment of the diagnosis.

6.4. Assessment of the stability of retention through partitioning

After completion of the column simulation, soil samples were also intended to assess the partitioning of heavy metals at different depths thanks to a sequential extraction protocol [20].

The partitioning was determined following the protocol in 4 steps developed by the BCR (Bureau Communautaire de Référence) for soils and sediments [21]: 1) metals of the exchangeable and acid extractable fraction (linked to carbonates notably); 2) metals of the reducible fraction (linked to iron and manganese oxy/hydroxides); 3) metals of the oxidizable fraction (linked to organic matters and sulphides); 4) metals of the residual fraction (the crystalline structure). This protocol was applied to soil sample (slices) at –1 cm; -2 cm; -20 cm and -45 cm. As the total content at –20 cm and –45 cm remains unchanged compared to the initial state, these levels represent the undisturbed state of soils. In those deep levels, depending on the metal (Cr, Cu, Pb Zn), the residual fraction represents from 45% to more than 90% of the total content of metal of interest; and the sum of the residual plus the oxidizable fractions represents from 75% to almost 100% of the total metal content. In addition, the exchangeable fraction, which never exceeds 50% of the total content in the first centimetre, decreases quickly with depth (all data merged): from the second centimetre layer, this fraction does not exceed 30% of the total metal in question and is more often below 20% (all data merged). Figure 3 illustrates the final partitioning of Pb and Zn after for the silty soil.

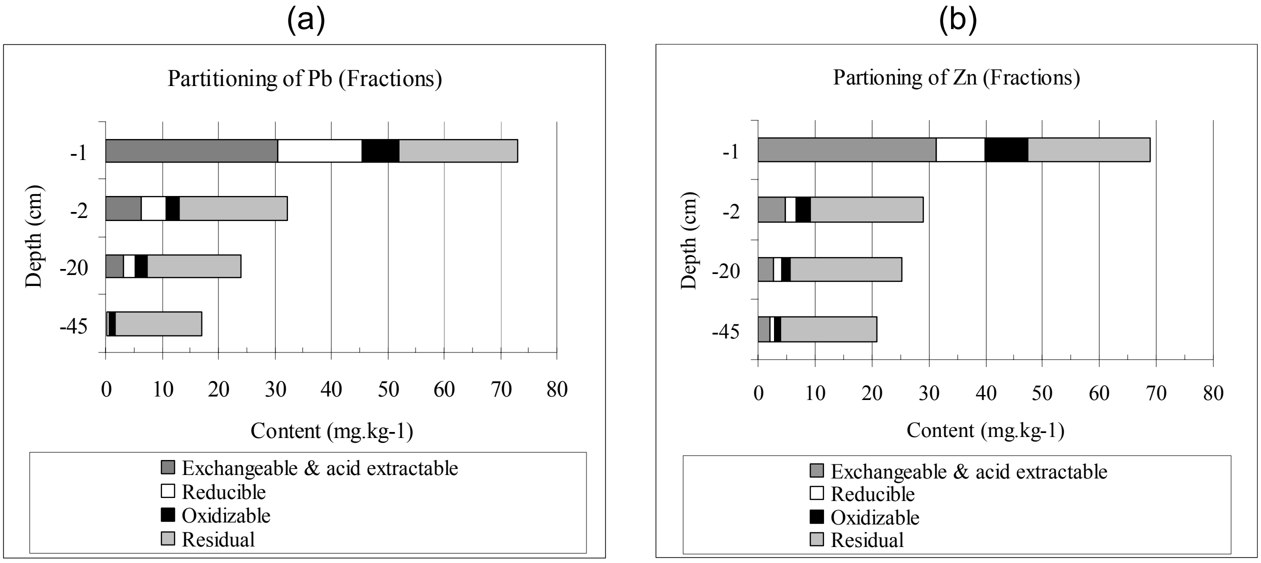

Figure 3 – Partitioning of Pb (a) and Zn (b) for the silty soil

These results highlight that even in the first centimetre of road soil, most of the heavy metal content will not be easily exchangeable (cumulated steps 2 to 4). The part of the exchangeable fraction (step 1) decreases fast from the second centimetre; partitioning indicates that in these upper levels the presence of metals is bound to the precipitation of calcium carbonate when the leachate (in which Portlandite dissolution induced a high Ca concentration: 634 $mg.l^{-1}$) enters in contact with the lower pH of the road soil.

7. LESSONS FROM COMPARISON TO REFERENCE VALUES

The normative approach such as the one developed in the Netherlands through the SPA [16] is useful to develop an harmonised policy of soil protection on a territory where various contaminant sources exist. The same definition of pollution (limit values) applies regardless of the source (polluting material or activity); this approach thus fully applies to the context of construction with alternative materials. Despite measured contents on-site or after column simulations represent rather unfavourable conditions (un-weathered and very alkaline MSWI bottom ash inducing a high Pb solubility in the laboratory case), they are well below the threshold values calculated according to the SPA methodology (ILV). Measured contents can be considered as fully acceptable from the point of view of a national reference system.

However, as contents are far below acceptability threshold values, the interpretation of the actual situation of the underlying soil can/must be more detailed thanks to additional comparisons. This is also justifiable because the road soil is not considered as polluted. According to this approach, the use of soil by road construction can be analysed under the same angle as other uses of soils. The subsequent effects in terms of contamination can be compared to the effects of a variety of other human activities. In France, the INRA research institute [11] has gathered such data for 237 sites and provides statistics for total contents for 9 metals of different soil textures (sandy, silty, clayey …). The positive conclusions of such a comparison were presented, however no data specific to road soil is compiled in this inventory, at first because of the scarcity of such data. But the fact that before to become a road soil, soil is first stripped in order to remove its organic and clayey horizons, then compacted [22], represents also a very special feature regarding the ability to fix pollutants. Road soil represents a particular case of soil use which limits the comparison with statistics on ordinary soils.

The low levels of contamination observed in comparison to the scarce normative and statistical references for soils submitted to human activities, gives ground for the development of the thinking about a tool suitable for the case of road soils. This tool must allow to go further than the single description of the *state* of soils; it must allow to highlight the *effects* specifically induced by the alternative material application. Beyond this, from the practical point of view, decision makers and engineers need tools that allow comparisons between alternative materials, road layer applications or structures, and natures of road soils.

8. DEVELOPMENT OF A SPECIFIC ASSESSMENT METHOD

As roads are built in an environment open to various sources of contaminants, in order to be able to assess the real contribution of the road made of alternative materials to the final state of road soils, the development of a specific assessment approach must integrate the quantification of actual road effects.

8.1. Description of the road system

After rainfall, the pollution induced by road traffic, maintenance or accidents, can infiltrate into the road body, and road layers can possibly release polluting agents toward road soil. The latter can also contaminate groundwater or be contaminated by it (if polluted). The road territory (bounded by the area of wet emissions generated by the traffic around the road structure) is also under the influence of the general pollution induced by various sources (domestic, industrial, agricultural), should it be carried through the atmosphere or through groundwater. In such a system, the relative state of road soil can be determined by comparison between inside and outside the road territory.

In order to isolate the effect of the road made of alternative materials, the reference state of soil must be found outside the road territory (i.e. outside wet emission influence). In order to prevent differential effects from contamination by specific pollution sources, and to avoid horizontal changes of the soil nature, the reference soil must be as close as possible from the boundary of the system. According to this approach, effects on the road soil result from the contribution of road use pollution and road structure. Figure 4 illustrates the setting of reference soil with consideration of the main pathways of soil contaminants.

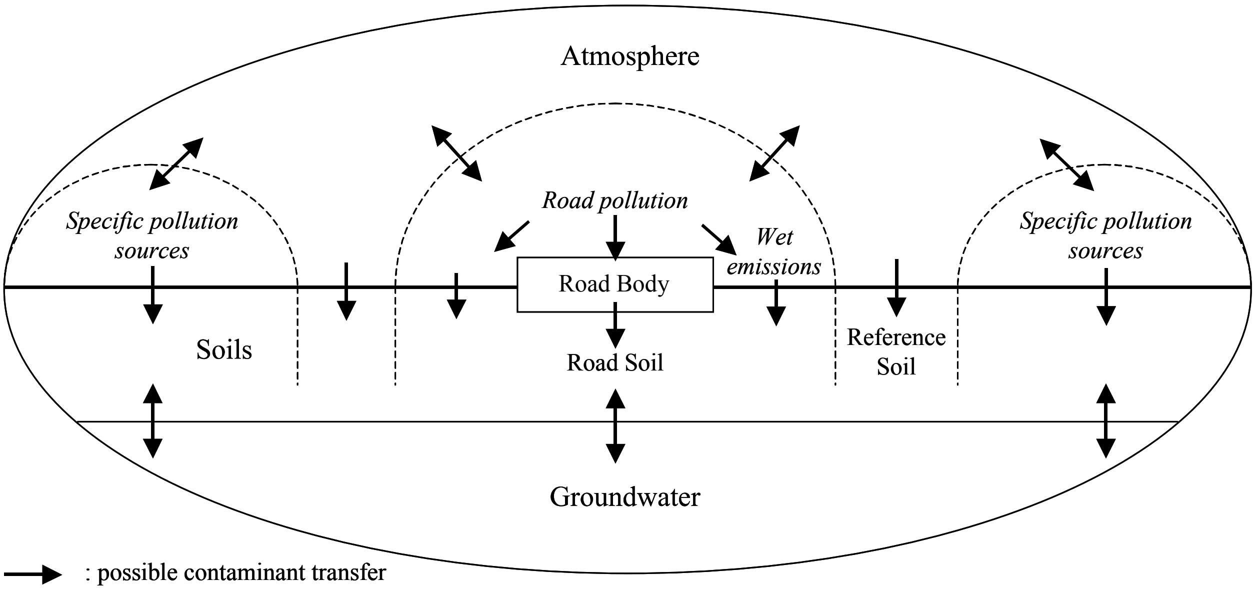

Figure 4 – Main pathways of soil contamination

8.2. Development of indicators for soil impact assessment

The impact assessment of road soils is obtained from a comparison between contents in the road soil at different depths and those of reference soil at the same depths [23]. The comparison is performed between vertical profiles for various parameters (C): contaminant contents; physico-chemical parameters such as pH, Eh, electrical conductivity... The depth at which the alternative material layer is in contact with the road soil is set as the vertical scale origin for all calculations (Figure 5). Each parameter value at the origin depth in the reference soil is labelled C_{origin}.

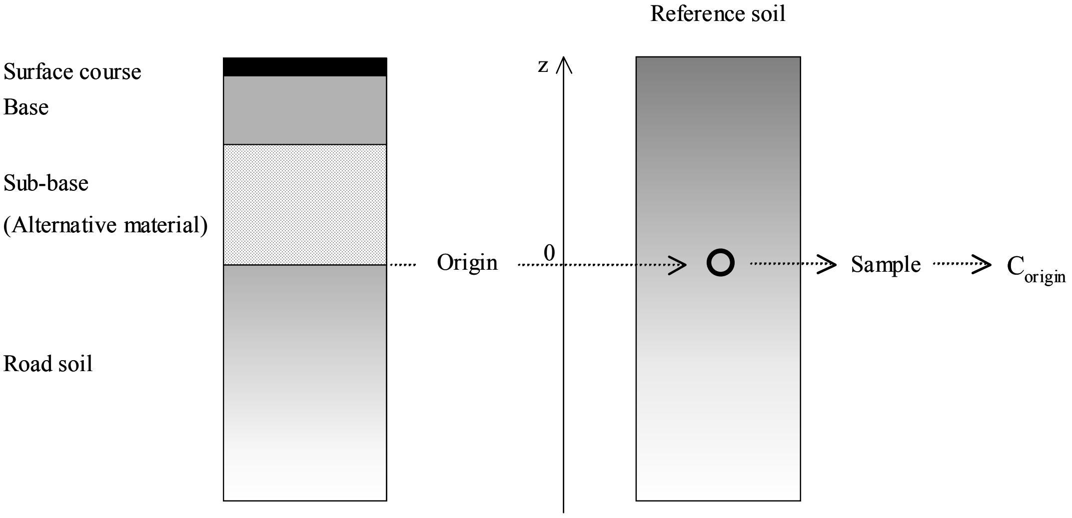

Figure 5 – Definition of the origin

For any parameter from the reference soil and the road soil, at any depth (d), the *Soil Indicator* (IS) which is not physically dimensioned, is expressed as:

$$IS(d) = (C(d) - C_{origin}) / C_{origin} \qquad (1)$$

Then, at each depth (d), the algebraic difference between IS values of road soil (RS) and reference soil (Ref) is investigated in order to identify any perturbation. This difference is called *Road Effect* (RE):

$$RE(d) = IS_{RS}(d) - IS_{Ref}(d) \qquad (2)$$

C_{origin} may be higher than C(d) measured elsewhere along the profile of the reference soil or of the road soil. In that case the corresponding IS(d) can range between –1 and 0. If this happens, this can induce also RE values comprised in the same range. As a consequence of this, IS and RE range is [-1; +∞]: the small negative range simply indicates the absence of effect. For a given site, comparisons are done between the same levels of soil (i.e. same nature). Thanks to this, the heterogeneity between sites regarding the influence of soil nature is cancelled out. Hence, the RE indicator formula can be used for a quantitative comparison between sites.

The concept of reference soil was used for the diagnosis studies in order to compare the general *state* of soils of each site (SA and SB) [12] but measured values were not used to apply the method to *effect* assessment [23]. The interest of such indicators is illustrated below. Results can highlight specific storage zones for some chemical elements. The magnitude of RE depends on chemical parameters and soil types. RE has also the ability to erase the misleading effect of some local environment conditions on observations. For both sites (SA and SB), RE profiles show the similar behaviour of Al and Cr, linked to the known similar behaviour of Cr_{III} and Al_{III} in response to changes in soil pH [10]. This example highlights similar RE profiles between different chemical species (Figure 6a).

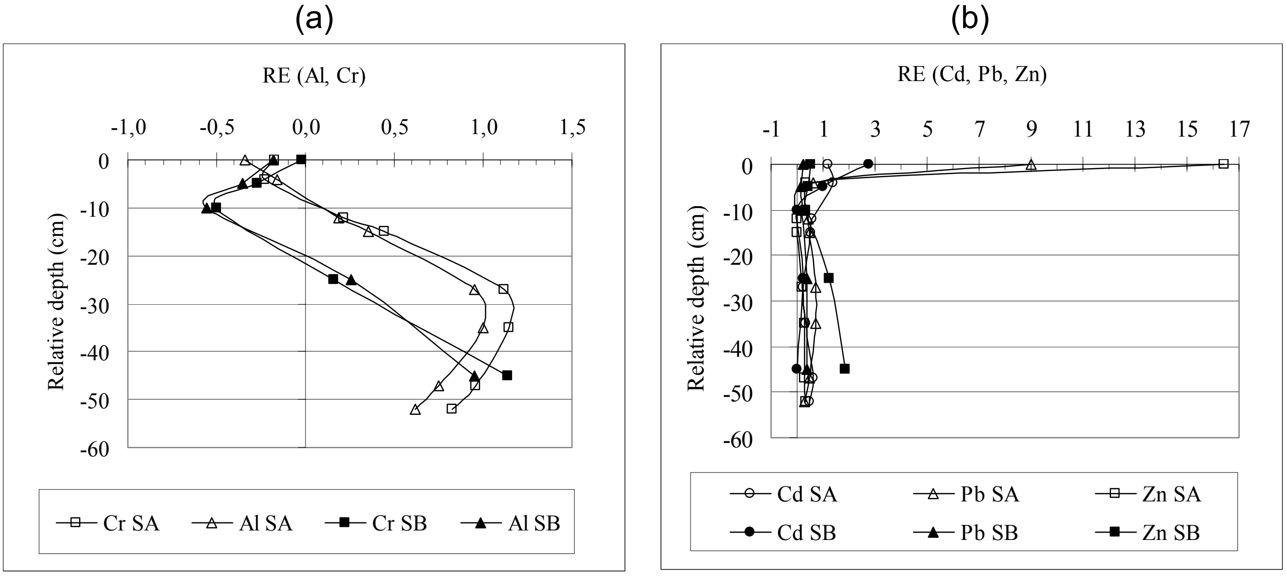

Figure 6 – Re profiles for Cr and Al (a) and Re profiles for Cd, Pb and Zn (b)

Cadmium, lead and zinc are pollutants that are often investigated regarding MSWI bottom ash. The RE for Cd is always below 1.5 in Site A (Figure 6b). In Site B, it is close to 3 at the origin. In both sites it decreases quickly. The RE for Pb is particularly high (9) at Site A origin; below -5 cm and all along the soil depth, it remains close to 0.5. At Site B, the Pb RE profile is vertical and equal to 0. The RE profile for Zn for both sites is similar to the one of Pb: very high (close to 17) at Site A origin, and equal to 0 at all other levels; equal to 0 all along soil depth at Site B.

8.3. Relevance, robustness and reliability of indicators

RE indicates rather well the actual road effect in contrasted environments. As an example, Site A reference soil profile showed an increase of chloride content with depth which was probably due to the proximity of the sea and to the subsequent groundwater salinity increase [23]. Easy fluctuations of the water table in the local littoral sand can have affected the local soil. Such a phenomenon was not possible in the continental context of Site B: the chloride content profile in the reference soil was vertical and contents almost nil. Now, finally, both sites RE profiles were similar in shape and magnitude.

RE variations with depth differ significantly from one road site to another and depend upon chemical species. Thus, despite higher RE are noticed at the origin level (RE(0) = 17 for Zn; RE(0) = 9 for Pb), for some elements, the RE is not the maximum at the contact between the road soil and the deeper road layer (see Figure 6a). The assessment by means of RE allows to prevent misunderstanding due to the influence of the local context. These indicators can serve to highlight ambient contamination and its possible interference with the actual contribution of road and materials; they allow discrimination between sources; the RE indicator allows for quantitative comparison between contrasted sites in terms of environment, road structures, soils, and road materials.

In view of the development of the method, the function of the road soil as a structural element of the road body (namely subgrade) must be considered the same way as it would be for another layer of the structure. The thickness over which the road soil is compacted corresponds to the thickness of the subgrade. The possible confinement of pollutants released from the alternative material into that layer must be considered in the definition of acceptance criteria. Additionally, as a possible consequence of the compaction and reduced permeability of road soils (compared to natural soils), the horizontal transfer of leachate above the former toward the shoulders of the road structure, or toward the lower points of the longitudinal profile of the road [24], should be considered in further work for the selection of sampling points and implementation of IS and RE.

9. CONCLUSION

Within the framework of alternative material environmental assessment for road construction, road soil is a target that up until now has remained relatively un-addressed. Field and laboratory studies show and explain the capacity of road soil to serve as a chemical barrier toward the migration of some heavy metals released by alternative materials, and consequently their role in the prevention of pollutant dispersion in the environment. The stability of fixing is observed until the medium term, but will have to be studied in more detail for longer term prediction. Compared to pollution reference values, the fixing of heavy metals into road soils seems acceptable. RE indicators allow a global appraisal of the actual effect on the road soil of structures made of alternative material. As they are based on a relative effect, they can be interpreted without consideration for acceptability limit values, which them deal with absolute contents. However, the interest for decision making to define threshold values for RE magnitudes exists, and further developments in that direction is needed. For the application of the method, the alternative material road layer was considered as the predominant source of contaminant. However, in view to define acceptable Road Effects, the “calibration” of RE indicators by means of data from road built with classical materials will be necessary. For the sustainable management of alternative materials and road soils, these developments must accompanied by the development of predictive laboratory methods regarding the leaching of alternative materials and the sorption on road soils.

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