



Pilot-scale road subbase made with granular material formulated with MSWI bottom ash and stabilized APC fly ash: Environmental impact assessment



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HIGHLIGHTS

- A granular material (GM) was formulated with MSW BA and a mortar (M).
- M was formulated with APC residues and Portland cement (50%/50%).
- Three field scale roads were built for environmentally assessing the use of GM.
- The mechanical properties of GM were improved by optimizing the mixing process.
- Based on the results at field, the encapsulating effect of cement is effective.

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ABSTRACT

A granular material (GM) to be used as road sub-base was formulated using 80% of weathered bottom ash (WBA) and 20% of mortar. The mortar was prepared separately and consisted in 50% APC and 50% of Portland cement. A pilot-scale study was carried on by constructing three roads in order to environmentally evaluate the performance of GM in a real scenario. By comparing the field results with those of the column experiments, the overestimations observed at laboratory scale can be explained by the potential mechanisms in which water enters into the road body and the pH of the media. An exception was observed in the case of Cu, whose concentration release at the test road was higher. The long-time of exposure at atmospheric conditions might have favoured oxidation of organic matter and therefore the leaching of this element. The results obtained showed that immobilization of all heavy metals and metalloids from APC is achieved by the pozzolanic effect of the cement mortar. This is, to the knowledge of the authors, the only pilot scale study that is considering reutilization of APC as a safe way to disposal.

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1. Introduction

During 2011 in Spain, 1.1 million tonnes of municipal solid waste residues (MSW) were incinerated according to EUROSTAT. This number accounts for 4.8% of the total MSW generated in the same year. Based on the same statistics, an average increase of 16% in MSW production since 1998 led to a rise of 60% in the incineration rate. Hence, it seems that incineration have become a valuable alternative in what MSW management concerns. Even though a substantial waste reduction can be achieved by this technique, an average 30% of the total content remains as the two main residues:

bottom ash (BA) and air pollution control (APC) ash. Different stabilization and solidification methods for both residues have been reviewed elsewhere [1–3]. The reuse of BA in construction applications has been preferred over other techniques as it has proved to be a feasible choice owing to its well defined chemical composition typically based in aluminosilicate phases. On the other hand, APC ash is currently classified as a hazardous waste due to its high chloride and heavy metals content [4]. Nowadays is generally handled by landfill disposal after stabilized with Portland cement [5].

An early work studied the properties of a granular material obtained from crushed concrete formulated with both natural weathered bottom ash (WBA) and air pollution control fly ash (APC) as aggregates with Portland cement used as a binder for its reuse as a road sub-base material [1]. In this study, the mechanical properties were characterized by unconfined compressive strength

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and abrasion resistance. At the light of the results obtained, the granular material was not adequate for its use as road sub-base material owing to the poor abrasion resistance observed, according to standard crushing Los Angeles coefficient test [6]. This fact was attributed to the high residues-to-cement ratio and, mainly, the high proportion of fines from APC. In what environmental behaviour concerns, the concentration of heavy metals and metalloids in the leachates [7] was below the concentration limits established by the Catalan legislation for the revalorization of residues [8]. Therefore, the reutilization of the proposed material as a granular material substitute in which high mechanical properties are not required can be regarded as an environmentally friendly option in which the potential leaching of toxic species is substantially reduced.

However, the results obtained from the standardized laboratory tests under equilibrium conditions might differ from those extracted from real scenarios as some important factors are not taken into account when single leaching tests are carried out. These can be summarized as follows: (i) Temperature difference between laboratory and field conditions, (ii) hydrology and climatic factors, (iii) pH and redox conditions, (iv) time scale, (v) mass transport rate, (vi) ageing of material by carbonation and oxidation, (vii) water infiltration rate and mechanisms, (viii) degree of water saturation and (ix) the heterogeneity of a real subbase with strong edge effects [9–13]. Each condition and the characteristic parameters are closely related to each other in an interaction matrix previously studied by Bendz et al. [12].

Taking into account all the above mentioned, the aim of this study is to assess the performance of the previously reported granular material in a field scale road site. However, in order to reduce the high quantity of fines generated under compaction and improving the stabilization of APC by the solidification process, the preparation of the granular material was made differently from the above described material. The reduction of fines was pursued by increasing the binder effect and the contact between particles. This can be achieved by mixing APC and cement separately from BA to form a mortar. This mortar, properly crushed, was subsequently mixed with BA to form an aggregate material with the same proportions as the previously studied.

By this manner, not only a mechanical improvement is achieved but also the results obtained from a pilot-scale study in real conditions are highly valuable as they can be used to evaluate and unequivocally report innocuousness of the proposed material.

2. Materials and methods

2.1. Materials

Approximately 6 tonnes of WBA and 1 tonne of APC were collected from a single municipal solid waste incinerator facility located in Tarragona, Spain. The feed stream is mainly composed of household rubbish, with a small input from commercial sources. Each year, 35,000 tonnes of BA and 3000 tonnes of APC residues are produced in this facility. The former, after a suitable short-term natural weathering stabilization process, is currently mainly reused as a secondary building material while the latter is stabilized with Portland cement prior to landfill disposal. The WBA taken was previously homogenized in a conditioned plant where its quality was improved and the metals recovered before being left at the open for weathering for up to four months. APC fly ash was collected directly from the storage silos. Representative subsamples of about 50 kg were taken from both kinds of residues for physical and chemical characterization.

The granular material (GM) was formulated in accordance with a previous work [1] and consisted of 80% WBA and 20% of

Table 1
Chemical composition of WBA, M and GM determined by means of XRF.

Oxides	WBA (%)	M (%)	GM (%)
SiO ₂	43.3	21.7	39.15
CaO	16.9	45.3	26.59
Cl	0.14	9.08	0.89
Fe ₂ O ₃	14.1	1.65	16.89
Na ₂ O	7.58	4.44	3.71
Al ₂ O ₃	5.80	3.82	4.87
MgO	2.22	1.87	2.02
K ₂ O	1.11	2.10	1.67
CuO	0.23	–	0.13
SO ₃	0.65	4.53	1.53
TiO ₂	0.35	0.68	0.3
P ₂ O ₅	1.97	0.51	1.1
ZnO	0.18	0.31	0.19
PbO		0.13	0.03

mortar (M). As previously mentioned, M was prepared separately and it consisted in 50% APC and 50% of Portland cement. A concrete admixture named *Mortar R* (Grace Construction Products) was also added and it consisted in a fluidizer polymer-based aqueous solution. The Portland cement used was of the type CEM I 52.5 R. The total quantity of M necessary for the two pilot scale road sites was calculated according to the volume of each subbase layer and taking into account the Proctor density of WBA (1700 kg m⁻³). The M was prepared using a cement mixer with a total capacity of 7 m³ and two batches were needed to prepare the whole material. Based on this, the corresponding mass of APC and cement needed to get a 50–50% mixture was prepared and weighted separately. The admixture dosage of 0.5% of the cement content was used as previous tests showed it was the optimum amount. Water-to-cement ratio was fixed in 0.8 for each mix as it was previously corroborated to give desired consistency in the range of 5–8 cm according to the Abrams cone method [14]. Fresh M was poured in two 1.5 m × 1.5 m × 0.5 m formworks and left for curing for 28 days. Six prismatic specimens with dimensions of 40 mm × 40 mm × 160 mm were cast separately for chemical and mechanical characterization at laboratory. Hardened mortar was removed from the formworks and crushed with a digger. The crushed mortar obtained was 0–30 mm sieved and then mixed with WBA to form the desirable formula of the GM (20–80%, respectively). Samples of both the mortar and the GM were taken for physical and chemical characterization and to evaluate the leaching behaviour according to dynamic batch and column leaching tests [7,15].

2.2. Characterization of materials

The chemical composition of the major and minor elements in the bulk WBA, M and GM was determined in duplicate by X-ray Fluorescence Spectroscopy (XRF) using a Philips PW2400 X-ray sequential spectrophotometer. The results are shown in Table 1. As previously reported in the literature, WBA is mainly composed of amorphous and vitreous phases, with Si being the most abundant constituent coming from lime-soda glass, followed by Ca from ceramic materials [11]. As for the M, its content was characteristic of cementitious materials with a high content of chlorides and calcium from APC generated during the gas cleaning process. As expected, the GM presented a content of CaO, SiO₂, SO₃, K₂O and Na₂O ranged between both materials as a consequence of the dilution effect by adding more WBA. The crystalline phases were determined by X-ray diffraction (XRD) in a Bragg-Brentano Siemens D-500 powder diffractometer with CuK α radiation. The results for WBA and GM (not shown) proved that amorphous SiO₂ and CaCO₃ are the main forms of occurrence. As for the M, taking into account that the hydrated phases of Portland cement are chiefly amorphous, it can be stated that the majority of peaks should be attributed to

Table 2
Los Angeles coefficient (LA) for 100% WBA and GM material. The values for typical natural aggregates are also presented for comparison.

Material	LA (%)
Weathered bottom ash (WBA)	33
Granular material (GM)	38
Natural aggregates	
Hard igneous rocks	10
Soft limestones and sandstones	60
Basalt	10–17
Dolomite	18–30
Gneiss	33–57
Granite	27–49
Limestone	19–30
Quartzite	20–350

APC. The predominant phases are halite (NaCl) and sylvite (KCl) formed from the chloride content. Other soluble phases such as CaCO₃, Ca–Al–Si–O and Ca(ClO)₂ are attributed to the excess of lime injected during the scrubbing of the combustion gases and the air pollution control system [16].

The particle size distribution of the GM and the WBA (not shown) was made by sieving the samples according to the standard EN 933-2 [17]. Both distributions were very similar and close to the optimum distribution depicted by the optimum Fuller Curve [18]. There is a sand-like fraction below the ideal and a slight excess on the coarse particles. Therefore the material can be classified as 0–25 mm with good graduation [19].

Compressive strength was tested for each M specimen following EN 12390-3 [20]. The results obtained from replicates showed an average value of 14.5 ± 2.3 MPa. The values are substantially higher than those obtained in the previous study [1] as a consequence of the major aggregate to cement ratio: mortar formulated with an APC to Portland cement ratio of 50–50% vs. concrete formulated with an mixture of WBA (80%)+ APC (10%) as aggregates and Portland cement (10%). As it can be observed, the APC to Portland cement ratio is the same, however the effectiveness of the solidification process is greater in the formulations of M than that of concrete. Moreover, in order to evaluate the potential reutilization of the GM as a subbase material, the abrasion resistance was measured. This parameter is depicted by the Los Angeles (LA) coefficient [6] and it gives information about the ability of an aggregate material to maintain its physical integrity under compaction [4]. The results for GM and WBA are shown in Table 2. The values of typical natural aggregates are also presented for comparison [21]. As it can be seen, the fragmentation or abrasion resistance increases as the LA coefficient decreases and the result of WBA shows a similar capacity of that of Gneiss while GM presented larger percentage losses as a consequence of fines from APC, which occurs as in the particles of mortar. However, these were fewer than those obtained in the previous study [1], demonstrating the effectiveness of preparing the M separately before mixing with WBA.

The potential release of heavy metals and metalloids was evaluated at lab-scale by means of two types of dynamic leaching tests. The batch leaching test was performed following the standard EN 12457-2 [7]. This test is commonly used to simulate a period of several years [22]. The second type of leaching test used was the column-percolation test described in CEN 14405 [15]. The main advantage of this test is that it can resemble real conditions as the leaching of granular wastes is in some circumstances dictated by percolation processes [23]. Some modifications were made to the cited standardized test so that it approached the pilot-scale conditions. First, the size of the samples was not reduced, and therefore a particular column was selected for all experimental trials ($\varnothing = 150$ mm; $h = 400$ mm). Demineralised water was used as a leachant and was pumped from the bottom to fully saturate the

column. The column was prepared, filled and packed according to the standard procedure. The proper mass of material to be packed in was calculated according to the thickness of the subbase layer in the pilot road site (200 mm), the column dimensions and the WBA optimal dry density and humidity (an average of 17.1%) in the case of the GM. The moisture content measured immediately after sampling was taken into account and the Proctor humidity was reached by adding demineralised water. An inert and compacted material was used between support filters and tested material at the top and at the bottom for mechanical assistance and to avoid filter obstruction. A constant flow rate of 6 ± 0.5 mL min⁻¹ was chosen and different times were fixed for collecting eluate fractions (10, 20, 30, 45, 60, 120, 360, 480, 1440, 1560, 1800 and 2880 min).

The eluate fractions collected from both leaching tests were analyzed with respect different parameters. First of all the pH and conductivity were immediately measured. Afterwards each sample was passed through 45 µm polypropylene membrane filters, acidified by adding a few drops of HNO₃ and preserved in a fridge at 4 °C for subsequent analysis. The analysis was conducted with respect the content of heavy metals and the concentration of sulphates and chlorides in solution. Trace metals concentrations were determined by inductive coupled argon plasma mass spectrometry (ICP-MS). In order to avoid the chloride interference, As was reduced for analysis using a potassium iodide solution and determined by the same technique using hydride generation. The content of sulphates was determined by means of a colorimetric method using an Aquamate spectrophotometer. As for chlorides, their concentration was measured by selective ion electrode (ISE). Experimental trials for M and GM were performed in duplicate. Therefore, the results are presented as the average.

The batch leaching results for the M, GM and APC are presented in Table 3. The concentration release of heavy metals and metalloids for GM was below the threshold for non-hazardous materials established by the Catalan Government for Landfill disposal and the regulatory limit values for reutilization as secondary building material [8,24]. In contrast, APC must be catalogued as hazardous due to its high Pb concentration as well as for the elevated values of Zn and Ba observed. The encapsulating effect of cement over APC is very effective at the light of the results obtained for the leaching of the M, specially for Zn and Pb, whose concentration release was substantially diminished below the limits for categorization as inert.

The corresponding results and discussion of the column leaching test are presented further, where they are compared with the leachates collected on field.

2.3. Pilot-scale experimental

2.3.1. Pilot site description

The experimental site consisted in three roads fully embedded in two metre height esplanade for an easy water discharge by gravity. Each road measured approximately 3 m × 5 m × 0.6 m and the structure cross section was composed of the following components from top to bottom: (i) an outermost cover (width = 200 mm) formed by natural ballast with sizes < 40 mm; (ii) a sub-base composed by the granular material object of study (width = 200 mm); (iii) a layer of drainage grave < 10 mm covering a PVC pipe system for the collection of leachates and (iv) a waterproof layer formed by a geotextile membrane as a bottom liner. Fig. 1 shows a schematic layout of the pilot scale road constructed as well as a picture of the three road units built. Roads units were one metre separated from each other and the bottom of each of them was slightly sloped towards the centre to ease the leachates collection through the pipeline system. This was formed by a perforated tube placed over the geotextile membrane and covered by the pebbles gravel to protect it against obstruction. Leachates were conducted

Table 3
Results of the leaching test EN 12457-4 for different road sites materials. Results expressed in mg kg⁻¹.

Element	APC fly ash	Mortar (M)	Granular material (GM)	Limit values			
				Landfill ^b			Utilization ^a
				Inert	Non-hazardous	Hazardous	
As	<0.52	<0.10	<0.01	0.5	2	25	1
Ba	39.3	22.8	<0.01	20	100	300	–
Cd	<0.02	<0.02	<0.01	0.04	1	5	1
Cr _{total}	<0.04	<0.04	<0.01	0.5	10	70	5 ^c
Cu	0.79	0.15	<0.01	2	50	100	20
Hg	<0.04	<0.04	<0.01	0.01	0.2	2	0.2
Mo	2.52	0.06	<0.01	0.5	10	30	–
Ni	<1.32	<0.04	<0.14	0.4	10	40	5
Pb	47.3	0.86	0.40	0.5	10	50	5
Sb	<0.04	<0.04	0.47	0.06	0.7	5	–
Se	2.82	<1.00	<0.40	0.1	0.5	7	–
Zn	19.8	0.54	1.16	4	50	200	20

^a Catalanian order number 2181/13.3.1996 [8].

^b Catalanian order number 5370/30.4.2009, Decret 69/2009 [24].

^c Max. Cr(VI): 1 mg kg⁻¹.

to polyethylene tanks with a 300 L capacity. The leachates collection system and dynamics will be further discussed.

Each road unit was formed by a different sub-base composition. The first one was entirely composed of weathered bottom ash (WBA) and was consequently treated as a reference. The second and third roads contained 100% of the granular material (100% GM).

2.3.2. Leachates collection

One of the main drawbacks when manipulating leaching data in field is the liquid-to-solid (L/S) ratio, as accounting the quantity of water put in contact with the solid material might be difficult if it is dependent on the rain fall intensity. For this purpose the utilization of pluviometers might be useful, although uncertainty has to be assumed when calculating the water infiltrated per square metre. Alternatively, precipitation data can be calculated in base of annual statistics [25]. In order to monitor L/S more efficiently, an aluminium roof structure was placed to cover the three road sites. This prevented any contact with rainwater. A water distribution system was placed at the top of each road. It was a formed by a net of hoses uniformly distributed in parallel. The main hose was put in series with the water supply system and the entering water was counted using an electronic metre. By this manner, the exact quantity of water put in contact with the granular material was registered.

An irrigation schedule was established regularly along the entire duration of testing (110 days) according to the pluviometry of the zone. Irrigation took always place during the mornings, with an average ambient temperature of 20 ± 10 °C. In order to simulate real conditions (i.e. drought periods), irrigation was stopped from day 50 to 70. The leachates generated by the incoming water passing through each type of subbase layer were collected by the pipeline system placed at the bottom. At this point collection was mainly divided in two. On the one hand and for every section, a sample of the leachates generated each time the roads were irrigated was collected immediately at the exit of the pipe system and subsequently catalogued as one-off leachate. The rest was conducted to the polyethylene tanks for accumulation. These leachates were labelled as cumulative and were taken regularly from the top of the tank after mixing. For both types of collected leachates the pH and conductivity were measured before passed through 45 µm polypropylene membrane filters, acidified by adding a few drops of HNO₃ and preserved in a fridge at 4 °C for subsequent analysis. The analyses were the same as in the laboratory scale.

3. Results and discussion

Column leaching experiments were performed in duplicate and therefore the forthcoming results are presented as the average. Analogously, results for GM at pilot scale are expressed as the average of roads two and three. The cumulative L/S ratio of the column leaching test started at 0.01 L kg⁻¹ and reached an average of 2 L kg⁻¹ after 48 h. As for leaching in field, the final L/S ratio achieved at the end of the experimental test was determined to be close to 0.1 L kg⁻¹ after four months.

The comparison of data from column leaching experiments and field can be carried on by expressing concentration release as a function of the L/S [23,26]. The pH values obtained at laboratory scale were in the range of 11.8–12.4 and 10.4–10.8 for GM and 100% WBA, respectively. On the other hand, the corresponding pH values in field remained in the 8–9 range during the whole experimental for the three test roads (100% WBA and the average for GM). In this aspect, Izquierdo et al. [27] have already highlighted the fluctuations of the pH values between EN 12457 laboratory and field tests for BA and those from the latter conditions have always been reported to be slightly lower. This can be explained by the uptake of atmospheric CO₂ during the storage of the leachates before sampling as well as during the entire four months. Likewise, previous studies have reported a temperature effect on pH and data spreading as a consequence [25]. In real conditions, especially during winter, temperature is expected to be lower than in laboratory and therefore might have an unexpected influence. Moreover, column leaching experiments were carried on without considering neither natural ballast nor drainage gravel, which may have influenced pH.

The conductivity values measured for the road sites containing GM were in the 20–40 mS cm⁻¹ range as a consequence of the presence of dissolved salts. These fluctuations are reported to be related to residence time and preferential flow pathways [25]. Otherwise, the corresponding values obtained at laboratory scale remained steady around 6 mS cm⁻¹. In this aspect, the main species responsible for conductivity from a qualitative point of view are mainly chlorides and sulphates, which are balanced by Na, K and Mg except at very high concentration values (the corresponding to the lowest L/S ratios), where Ca is also involved in compensating chlorides [28,29].

The release patterns of Cl⁻ and SO₄²⁻ for the GM road sites and the corresponding column experiments are presented in Fig. 2a and 2b respectively, as a function of the L/S ratio. For the sake of better comparison, each graphic includes an internal image showing a zoomed image of the earliest L/S values. The release of chlorides,

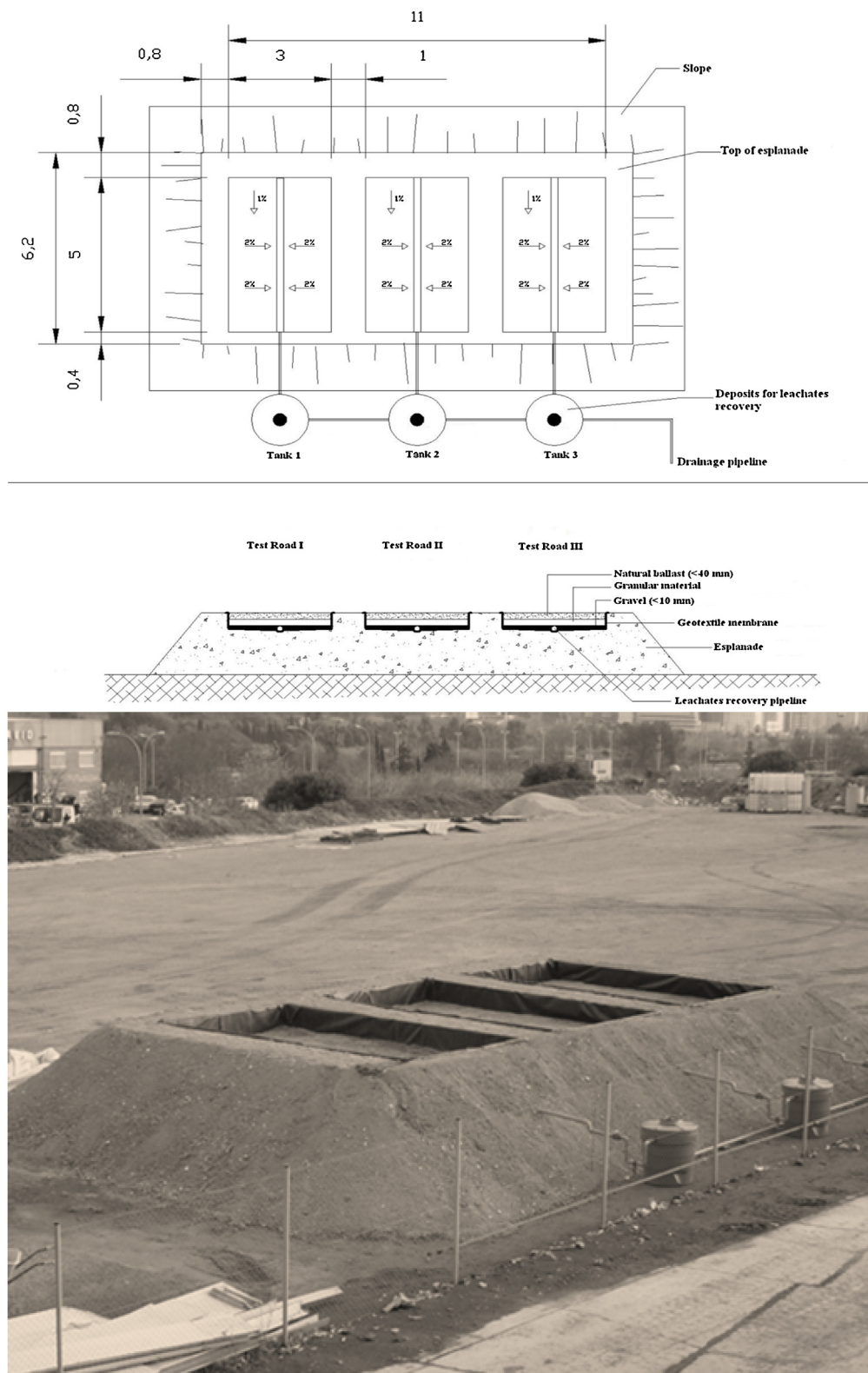


Fig. 1. Schematic layout of the pilot scale road built. Structure and measures of the three pilot scale roads built. Units of length in metres.

mainly from encapsulated APC, at both scales showed a wash-out peak which is typical of availability-controlled leaching as a consequence of their great solubility [3,30]. As the release of Cl^- is independent of pH variation [31], the difference between both scales is attributed to contact time between percolation water and

solid material for the same L/S ratio. Unlike column experiments, the solubility equilibrium of Cl^- at the test road was attained.

As for sulphate release, which is reported to be pH-dependent [31,32], steady values under 2000 mgL^{-1} were obtained for the road site and an increasing pattern of higher values was

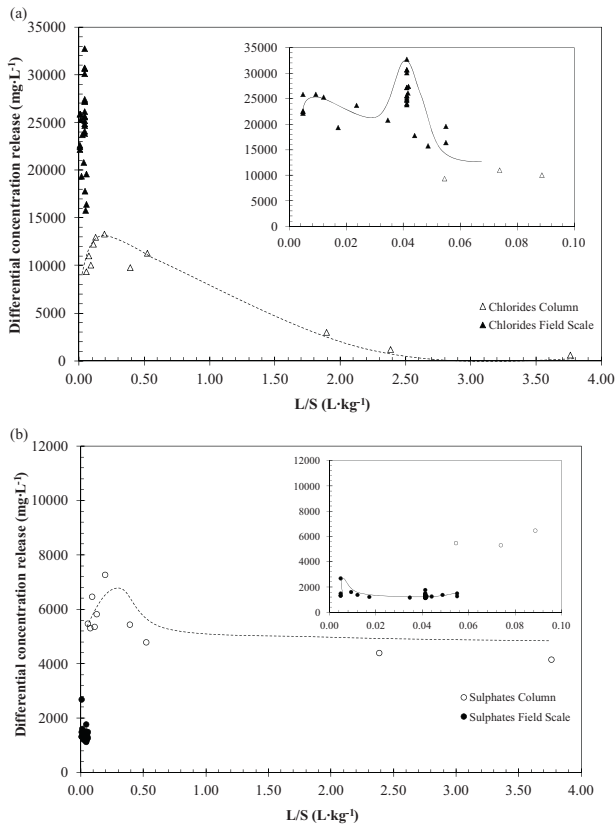


Fig. 2. (a) Concentration release of chlorides (mg L⁻¹) at laboratory and field scale as a function of the cumulative L/S for GM. (b) Concentration release of sulphates (mg L⁻¹) at laboratory and field scale as a function of the cumulative L/S for GM.

observed for column experiments. Contributing to pH, the leaching behaviour of sulphate is also dependent on other factors, including carbonation influence in the concentration of calcium, ageing of the BA sample, and formation of mineral phase such as ettringite and gypsum, which means that a substantial influence of the atmospheric open conditions at the road site might explained these differences [33].

The average results of the differential concentration release of some heavy metals and metalloids for the one-off leachates as a function of the L/S ratio is presented in Fig. 3 for test roads number two and three (GM). The selected species were those included in the Catalan Legislation for the revalorization of residues: As, Zn, Ni, Pb, Cr and Cu. Cadmium and mercury, which are also included in

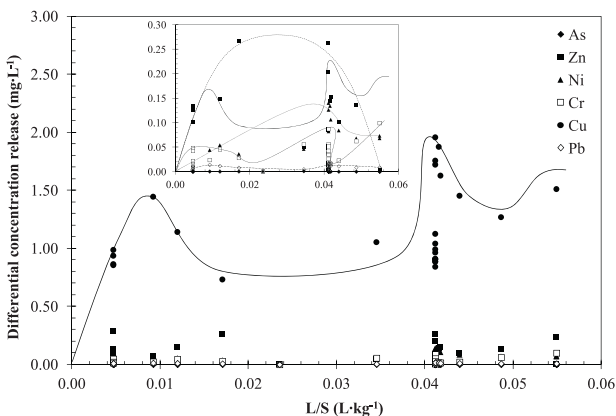


Fig. 3. Average differential concentration release of one-off leachates as a function of the L/S for test roads number 2 and 3.

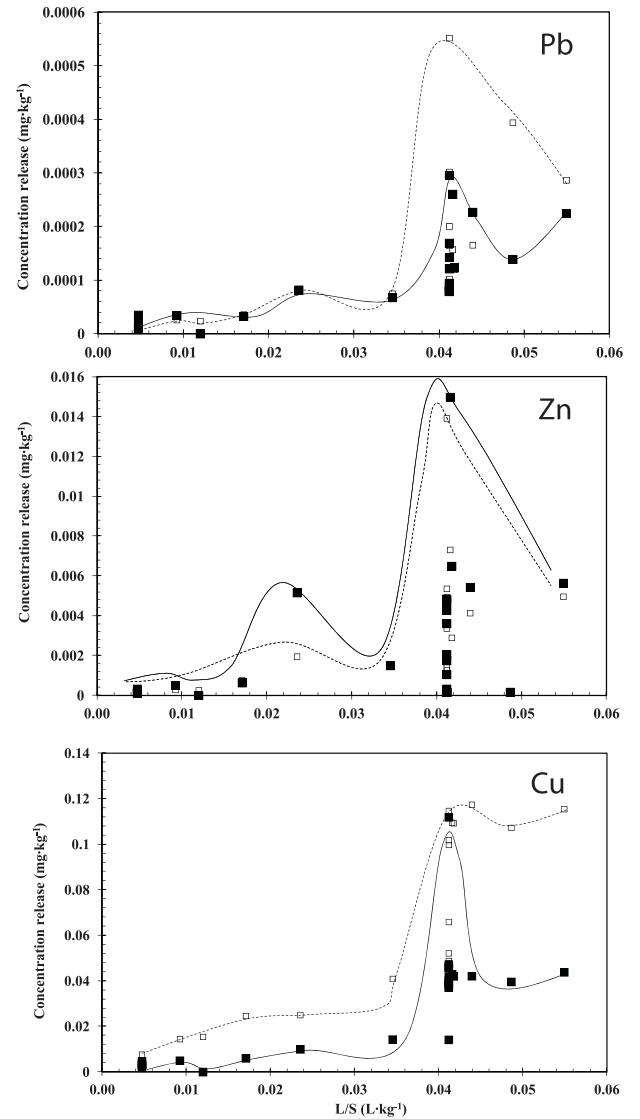


Fig. 4. Comparison between cumulative concentration release values of Pb, Cu and Zn as a function of L/S for WBA and GM. (■) WBA; (□) GM.

the aforementioned legislation, were below the detection limit of the technique (<0.10 mg L⁻¹) and thus their concentration release over time can be considered to be under the legal threshold. The release of Cu stood out the rest of elements and for better comprehension a smaller graphic is included as above. As it can be seen, all metals showed a concentration release increasing over time until stabilizing around a L/S ratio of 0.05 L kg⁻¹ and subsequently tending to diminish. The immobilization of the majority of these metals is attributed to the pozzolanic reactions promoted by the cement content [34].

A comprehensive analysis of the leaching behaviour can be deduced from the cumulative leachates. A comparison between the release pattern of the 100% WBA and GM road sites is presented in Fig. 4 as a function of L/S ratio. In order to assess the contribution of APC to leaching, those metals showing higher release were selected (Pb, Cu and Zn). As it can be deduced from this figure, the leaching of metals should not be attributed to exclusively APC and the stabilization/solidification of this residue was attained.

Fig. 5a and b compares pilot scale results of GM and the corresponding from the column leaching tests again for the cumulative leachates. Again, a zoomed image of the leaching at the early stages is presented for clearer comparison. In all cases except Cu, the

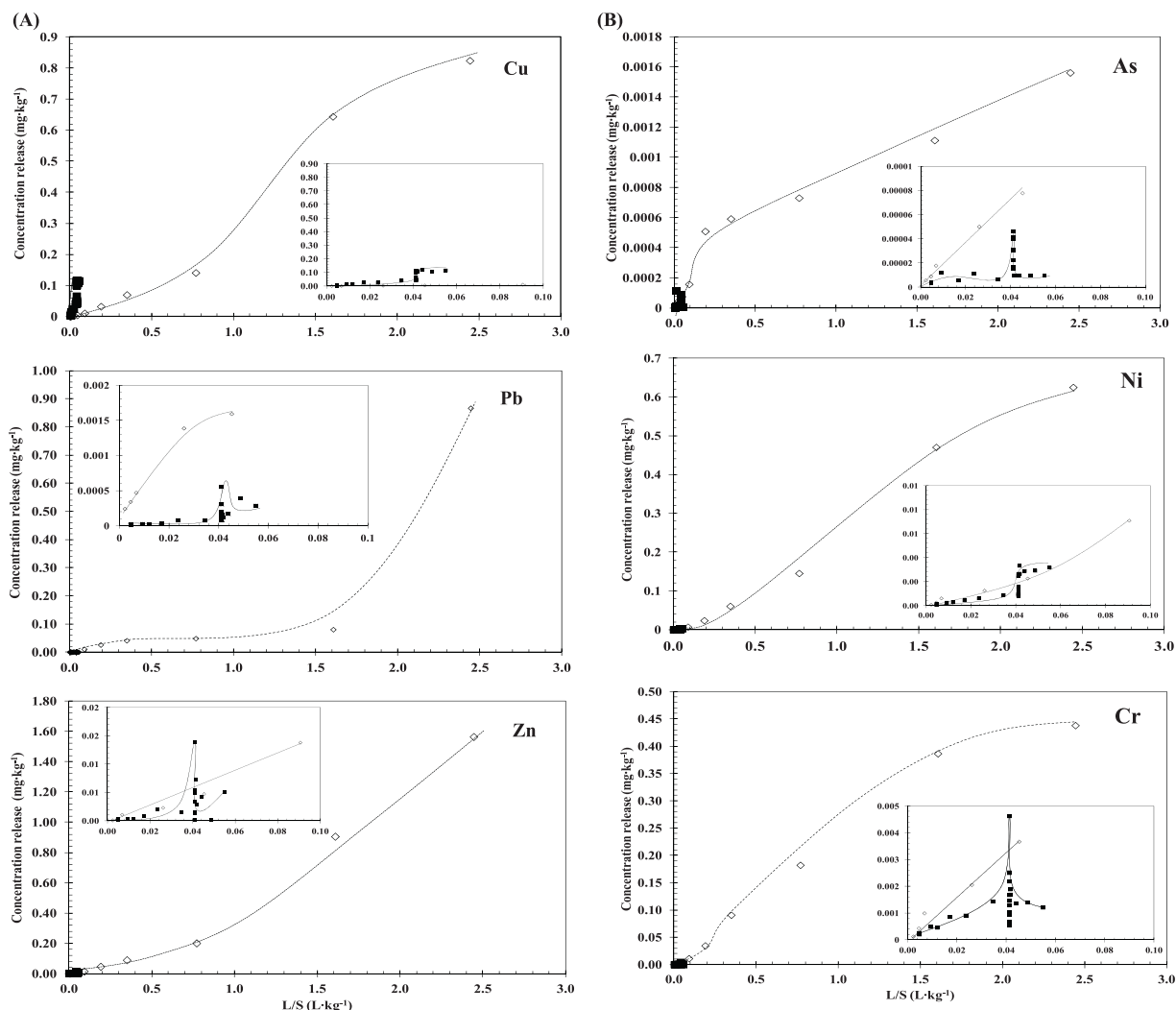


Fig. 5. (a) Cumulative concentration release values of Pb, Cu and Zn as a function of L/S for GM at both scales. (■) Test Road; (◇) Column Test. (b) Cumulative concentration release values of As, Ni and Cr as a function of L/S for GM at both scales. (■) Test Road; (◇) Column Test.

release at pilot scale was lower than those values predicted by the column leaching test. The overestimations in a column regime might be explained by the potential mechanisms in which water enters into the road body [35]. The mass transport rate of leachates starts with the dissolution of species by inlet water and a subsequent diffusion mechanism through the road layers. Unlike column experiments, both driving forces can be affected by the preferential flows of inlet water as well as by water runoff, edge effects and lateral inputs, easing disparity between laboratory and field scale [30,36].

As it has been previously remarked, L/S was low for the present study. Apart from pH, this ratio strongly affects the leaching behaviour, especially for those processes that are solubility controlled. At large scales, L/S is conditioned by climatic conditions, hydrology and hydrogeology of the site as it can be seen in the sharp release for all metals at L/S of 0.04 L kg^{-1} , coinciding with the stop of irrigation (simulating drought periods) at summer season and highest temperatures. During the simulated drought period, each road unit continued to release small volumes of leachates, whose corresponding L/S ratio was almost kept constant (0.04 L kg^{-1}) while increasing concentration, resulting in a sharp peak. Even though the time period of investigation was shorter than other pilot scales [13,25], it can be inferred from the zoomed images for all metals that the decrease in the release after the concentration peak might

lead to a stabilization plateau. To support this, the pH range of the experimental test was that of minimum solubility for the majority of heavy metals and metalloids [37] and is expected to decrease over time by the effect of atmospheric carbonation, among others (i.e. iron oxidation), favouring immobilization.

In the case of Cu (Fig. 5a), the concentration release at the test road was higher than leaching at lab scale. Although dissolved organic carbon (DOC) was not measured, according to the literature, it is expected that Cu might be linked to the release of DOC, as it usually forms strong complexes with organic substances such as hydrophilic and hydrophobic acids [13,37,38]. Compared to the short time of performance at lab scale, the long-time of exposure at atmospheric conditions might have favoured oxidation of organic matter and therefore the leaching of this element. The subsequent immobilization of this metal can be attributed to the formation of a less soluble phase of copper (hydr)oxides when exposed to the road site pH or the sorption into amorphous Fe/Al-(hydr)oxides [37,39].

Pb (Fig. 5a) is considered to be the most toxic element present in the incineration residues because of its potential dangerous effect on the environment and to human health. Its behaviour is reported to be similar to Zn (Fig. 5a), with maximum solubility at pH values over 11 [30]. Owing to the abovementioned pH range, their immobilization over time is assured.

Arsenic (Fig. 5b) is an oxyanion-forming metal and like Pb deserves special attention due to its toxicity. The release of As below the legislation limits was slightly higher for the GM than 100%WBA as its speciation might be enhanced by the Cl^- ions [40] from APC. Adsorption of arsenates over Fe (hydr)oxide surfaces is attributed to be the main mechanism controlling the leaching of this metal [37,39].

Nickel (Fig. 5b), like other divalent metals, is also pH-dependent and its release is mainly attributed to the low solubility of $\text{Ni}(\text{OH})_2$ in the test pH, which is reported to be the main equilibrium controlling phase [39]. Furthermore, as the pH decreased by the effect of carbonation, sorption into neoformed iron (hydr)oxides will also takes place.

In the particular case of Cr (Fig. 5b), its concentration in the available states of oxidation is strongly influenced by pH. In alkaline conditions the oxidation of Cr^{3+} to the more mobile Cr^{6+} is favoured and high redox potential values are required [3,41]. As abovementioned, the amount of Cr leached gradually increased over time, intensifying at L/S of 0.04 L kg^{-1} and subsequently reaching the stabilization plateau.

4. Conclusions

Pilot-scale studies are mandatory in the environmental science research field as they can undoubtedly report results that take into account the real factors and parameters of a given study. Laboratory scale experiments must be the starting point in this aspect as they have the ability to report the feasibility of a certain material to be reutilized in a safe way to the environment and human health. In this study two objectives were attained. The first one intended to prove the feasibility of using a granular material, formed with both WBA and a mortar formulated with APC and Portland cement, to be used as a subbase material. This feasibility was assessed from an environmental point of view as well as by establishing the minimum mechanical properties required to put it in service. The potential leaching of heavy metals, metalloids, chlorides and sulphates was assessed by monitoring them during 4 months with artificial irrigation. Even though the time of investigation was shorter than other studies reported in the literature, the results obtained showed that immobilization of all heavy metals and metalloids from APC is achieved by the pozzolanic effect of the cement mortar. Moreover, the absence of an asphalt cover might have overestimated the leaching results. Secondly, this is, to the knowledge of the authors, the only pilot scale study that is considering reutilization of APC as a safe way to disposal and therefore subjected to the Spanish patent number 201330986 in forced since July 2013 [42].

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