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REMOVAL OF HEAVY METALS USING DIFFERENT POLYMER MATRIXES AS SUPPORT FOR BACTERIAL IMMOBILISATION

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Capsule

Immobilisation of bacteria in the naturally occurring alginate and pectate and in a synthetic cross-linked polymer increased the Zn and Cd removal abilities from single and binary contaminated waters; the applications with the synthetic polymer were the most promising for Cd and Zn removal in single and binary mixtures.

Abstract

Great attention is focused on the microbial treatment of metal contaminated environments. Three bacterial strains, 1C2, 1ZP4 and EC30, belonging to genera *Cupriavidus*, *Sphingobacterium* and *Alcaligenes*, respectively, showing high tolerance to Zn and Cd, up to concentrations of 1000 ppm, were isolated from a contaminated area in Northern Portugal. Their contribution to Zn and Cd removal from aqueous streams using immobilised alginate, pectate and a synthetic cross-linked polymer was assessed. In most cases, matrices with immobilised bacteria showed better metal removal than the non-inoculated material alone. For the immobilisation with all the polymers, 1C2 was the strain that increased the removal of Zn the most, whereas EC30 was the most promising for Cd removal, especially when combined with the synthetic polymer with up to a ca. 11-fold increase in metal removal when compared to the polymer alone. Removal of individual metals from binary mixtures showed that there

was differential immobilisation. There was greater removal of Cd than Zn (removals up to 40 % higher than those showed for Zn)..

Keywords

Heavy metals, bacterial immobilisation, synthetic polymer, acetate, pectate, binary mixtures

Introduction

Heavy metal pollution is one of the most important environmental problems today, especially in relation to water contamination. Several industries, mining and smelting, as well as production of fuel, energy, fertilizers, metallurgy, electroplating, electrolysis, leatherworking and photography [1] produce waste and wastewaters that are discharged in water courses threatening the ecosystems and ultimately human health. Traditional methods of metal removal generally consist of physical and/or chemical approaches which are often expensive, with high energy and chemical requirements, producing high amounts of residues [2]. They are often not effective especially for low to moderate metal concentrations [3]. In this context, the search for more effective methods is necessary to reduce heavy metal contamination in waste water to environmentally acceptable levels. Biologically-based, eco-friendly and economically more attractive technologies are required.

Biosorption is a method that involves the use of biological materials that form complexes with metal ions using their functional groups [4]. In the process, a chemical link between functional groups on the biosorbent and the metal ions present in solution or an ion-exchange reaction due to the high ion-exchange capacity of the biosorbent may occur [5]. Bacteria have a high surface area-to-volume ratio and can thus provide a large contact surface, which allows the interaction with metals in its surroundings [6], and have been successfully used as biosorbents [7, 8, 9]. However, studies demonstrate that sometimes living systems are inconsistent, especially when using freely suspended biomass. In fact, although freely suspended biomass can promote higher contact with the contaminants during the removal process, it is usually unpractical as a clean-up method [10]. Biopolymers are non-toxic and when used to immobilise biomass may help improve biosorption capacity and facilitate biomass separation from metal bearing solutions. This can then be a non-destructive process if necessary and allow the

regeneration of biosorbents for multiple uses, as well as increasing biomass concentration [11, 12]. The ion-exchange process that occurs in such polymers when exposed to water contaminated with metals [13] is complemented with the biosportion capacity of the immobilised microorganisms. Other alternative is the use of synthetic polymers as matrices that can control or promote bio-adhesion. Potential applications for materials that are bio-adherent or bio-compatible are widespread [14]. Usually the synthesis of functional polymeric materials involves the use of a functional monomer to impart the desired characteristics to the final material and a cross-linker which will give the necessary rigidity to the polymer network. The main advantages of using these materials is the possibility to fine-tune the final properties by varying polymer composition, robustness and stability under a wide range of chemical and physical conditions.

Common matrices used to support organisms (either of natural or synthetic origin) include hydrogels [15], activated alumina and charcoal [16], kaolin [2], polyacrylonitrile [17], alginate and pectate.

The objectives of this study were to compare the use of alginate, pectate and a synthetic porous cross-linked polymer as immobilisation matrices for metal resistant bacteria species, and to evaluate the effect of the application of different bacteria in the removal of the metals Cd and Zn alone and as mixed metal solutions from contaminated water.

Materials and Methods

Isolation and selection of heavy metal resistant bacterial strains

Selected bacterial species were isolated from a metal contaminated site - Estarreja, Northern Portugal. Despite the high presence of metals – average levels of 835 mg Pb kg^{-1}, 66 mg Hg kg^{-1}, 26 mg Cr kg^{-1}, 37 mg Ni kg^{-1}, 16 800 mg Fe kg^{-1} and 3620 mg Zn kg^{-1} (total Zn) – the area is prolific in vegetation [18]. Several bacterial strains were isolated from the non-rhizosphere and rhizosphere soils. Soil samples were collected and serially diluted in saline solution (0.85% (w/v) NaCl) and inoculated on trypticase soy agar (TSA; Oxoid) at 30 ºC. Visually different colonies selected on the basis of colony morphology and colour were further purified [19]. For this study, 3 strains isolated at pH 7 designated as 1ZP4, EC30 and 1C2, were selected based on their metal tolerance in *in vitro* screening assays. Cell morphology was tested as described by Alexander & Strete [20]. Gram staining tests were performed as described by Murray et

al. [21] and Smibert and Krieg [22]. The pH range for growth was determined in buffered trypticase soy broth (TSB) adjusted at pH 3-10 (at 1 pH unit intervals). The turbidity of the cultures grown in an orbital shaker at 25 ºC was measured at 610 nm. All buffer solutions used to adjust the pH of TSB were prepared from 1 M stock solutions [23]. Citrate buffer was used for pH 3-6, phosphate buffer for pH 7, Tris-HCl buffer for pH 8, and a carbonate-bicarbonate buffer for pH 9 and 10. Growth temperature ranges were determined at 15, 20, 25, 30, 37 ºC on TSB and on TSA at 4, 10, and 50 ºC. Extraction of genomic DNA, PCR amplification of the 16S rRNA gene and sequencing of the purified PCR products were carried out as described by Rainey et al. [24]. Cloning of the amplicons into pGEM T-Easy vector (Promega) and cycle-sequencing were performed at Macrogen Inc. (Seoul, Republic of Korea), using 16S universal bacterial primers (f27, f518, r800, r1492) [25]. The quality of the 16S rRNA gene sequences was checked manually by the use of the BioEdit program (version 7.0.5.3) [26], and the sequences were aligned against representative reference sequences of the most closely related members obtained from the National Center for Biotechnology Information database [27].

Effect of metals on bacterial growth in suspension cultures

300 ml Erlenmeyer flasks containing 100 ml TSB supplemented with heavy metals at concentrations of 50, 100 mg l^{-1} (Cd^{2+}), 100, 250 mg l^{-1} (Zn^{2+}) and metal mixtures of 200 mg l^{-1} ([100 mg l^{-1} (Cd^{2+}) + 100 mg l^{-1} (Zn^{2+})]) were inoculated with the bacterial strains in order to achieve a starting optic density (OD) of 0.1 at 610 nm. The metals were applied as salts $ZnCl_2$ and $CdCl_2$. All the cultures, including controls (in triplicate), were incubated at 30 ºC for 24 h at 150 rpm. Bacterial growth was monitored at time intervals by measuring the optical density at 610 nm and the specific growth rate of each strain was determined. The strains with the highest growth rate were EC30, 1ZP4 and 1C2 and were selected for further characterisation and for the uptake tests.

Synthetic cross-linked polymer synthesis

Polymers were prepared by mixing in a 100 ml glass bottle 40 g ethylene glycol dimethacrylate, 0.37 g N, N-diethylamino ethyl methacrylate, 2 g polyethylene glycol 35000, 40.37 g N, N-dimethylformamide and 0.85 g 1,1'-azobis cyclohexanecarbonitrile. The mixture was bubbled with nitrogen for 5 min and sealed with teflon coated caps. Polymerisation took 20 min and was initiated using an

UVAPRINT 100 CVI UV source with a 0.163 W/cm^2 intensity [28]. The resulting polymer monolith was crushed manually in a mortar with a pestle and the particles in the range 200-500 μm collected using sieves from Endecotts, UK. Polymers were then washed with methanol overnight in a sohxlet apparatus in order to remove any unreacted monomers and the polyethylene glycol and after dried at 60 ºC during 6 hours. Polymers were produced with weak alkaline monomers in order to promote bacterial adhesion. The composition of the polymer was adapted from Barral et al. (2010) [29].

Bacterial Immobilisation

The bacterial strains (EC30, 1ZP4 and 1C2) were grown in 300 ml Erlenmeyer flasks containing 100 ml TSB until the cell biomass reached an OD of 1.0 (610 nm). Cells were harvested by centrifugation at 6000 rpm for 15 min and the bacterial pellet weighed and washed using sterile ultra-pure water. The harvested biomass was re-suspended in 25 ml sterile Universal bottles containing 5 ml of saline solution (0.85 % w/v).

For Ca-alginate and Ca-pectate, the bacterial inoculum was immobilised under aseptic conditions, using the method described by Escamilla et al. [30] and Montes and Magaña [31] with some modifications. The inoculum [OD=1 (610 nm), which represented a fresh weight of 74 mg for 1C2, 108 mg for 1ZP4 and 128 mg for EC30, in a volume of 100 ml] was adjusted in a volumetric cylinder to 1:1 inoculum/polymer ratio by using alginic acid (Sigma) or polygalacturonic (Sigma) 4 % (w/v) concentrated. The solution was homogenized and forced though a needle template (gauge for ± 3 mm beads) with a peristaltic pump (Watson-Marlow Bredel, Wilmington, Mass.) flowing at 10 ml m^{-1}, and the droplets were collected in a sterile gel inducer solution of 3.5 % (w/v) $CaCl_2$. After soaking for 1 h, the liquid was decanted and the spherical beads were washed with sterile ultra pure water. In aseptic conditions the beads were then packed into sterile 6 ml fritted SPE tubes (Supelco) with a filter. An adaptor cap (Phenomenex) was fitted to each of the tubes. For the synthetic polymer, 1 g was packed in sterile 6 ml fritted SPE tubes (Supelco) containing a filter under aseptic conditions. Bacterial biomass was then added to the tube (fresh weight of 150 mg). An adaptor cap (Phenomenex) was fitted to each of the tubes. Tubes were then left to settle for 1 h at room temperature. An additional alternative method was used with the synthetic polymer. The bacterial strains were grown in 300 ml Erlenmeyer flasks containing 100 ml TSB and 3 g of the

synthetic polymer until cells grew to 1.0 OD (610 nm). Cells and polymer were then harvested by centrifugation at 6000 rpm for 15 min and the bacterial and polymer pellet was weighted. Under aseptic conditions 1.5 g of the pellet containing the bacterial biomass and the synthetic polymer was packed in sterile 6 ml fritted SPE tubes (Supelco) with filter. An adaptor cap (Phenomenex) was fitted to each of the tubes. Tubes were then left to settle for 1 h at room temperature.

In every case, polymers were washed prior use and recirculation was made until OD of washing solution was bellow 0.1 (610 nm).

Heavy metal uptake tests

For metal uptake batch experiments, 5 ml of a solution (pH ranging from 6.50 to 7.01) containing 100 mg l^{-1} of Cd^{2+}, Zn^{2+} or a mixed metal solution containing 100 mg l^{-1} of each of the metals was added to the polymer packed tubes – metals for the solutions preparation were applied as their salts $ZnCl_2$ and $CdCl_2$. Three sequential cycles of 5 ml were tested for each treatment, with an average contact time of 2 min. Outlet solutions were collected filtered using a Puradisc 25 Syringe Filter (Whatman) and the amount of residual metal present in solution was measured by atomic absorption spectrophotometry in a Hitachi Z-8100 Atomic absorption spectrophotometer, with Zeeman correction.

Statistical analysis

Each treatment was comprised of 3 replicates. Statistical analysis was performed using the SPSS program (SPSS Inc., Chicago, IL Version 15.0). The data were analysed through variance analysis (ANOVA). To detect the statistical significance of differences ($P<0.05$) between means, the Tukey test was performed.

Results

Bacterial strains

The tested phenotypic characteristics of strains 1ZP4, EC30 and 1C2 are given in Table 1. The pH and temperature ranges for growth of the strains were similar. Full length (about 1250-1450 bp) 16S rRNA of strains 1ZP4, EC30 and 1C2 were sequenced and the closest affiliation according to sequencing were for strain 1ZP4 *Sphingobacterium* sp. MG2 (AY556417), for EC30 *Alcaligenes* sp. S-SL-5 (FJ529025) and for 1C2 *Cupriavidus* sp. 2CSa-12 (GU167923).

198

199 Growth of 1ZP4, EC30 and 1C2 in the presence of heavy metals

200 Growth curves for strains 1ZP4, EC30 and 1C2 in the presence of Zn^{2+} are shown in
201 Figure 1. At the concentrations tested, Zn^{2+} had only a small effect on their growth.
202 Growth of strains 1ZP4, EC30 and 1C2 was significantly reduced when TSB medium
203 contained Cd^{2+} (Figure 1). 1C2 was the strain most affected by the presence of Cd.
204 Remarkably, none of the tested strains showed a significant lag phase. Final biomass
205 concentration was lower when 100 mg l^{-1} of Cd^{2+} was applied (Figure 1).
206 When a metal mixture was used growth of strain 1C2 was visibly reduced (Figure 1),
207 which can possibly be attributed to the presence of Cd. On the other hand, the metal
208 mixture had less effect on the growth of strains EC30 and 1ZP4. In fact, for strain
209 EC30, part of the exponential growth phase was similar to the control growth (Figure
210 1).
211

212 Removal of single metals in solution by different matrices and immobilised bacterial
213 strains

214 *Removal of Zn*

215 The matrix type and bacterial immobilisation had a significant ($P<0.05$) effect on Zn
216 removal. In general, the treatments that included bacteria showed significantly ($P<0.05$)
217 better Zn removal than the matrices on their own, as shown by the significantly lower
218 concentrations of Zn in the outlet of the cartridges. ANOVA two way test results were,
219 in summary, after the first removal cycle, $F_{Zn(matrix)}=434$ ($P<0.001$), $F_{Zn(bacteria)}=1124$
220 ($P<0.001$) and $F_{Zn(matrix*bacteria)}=154$ ($P<0.001$); for the 2nd cycle $F_{Zn(matrix)}=446$
221 ($P<0.001$), $F_{Zn(bacteria)}=725$ ($P<0.001$) and $F_{Zn(matrix*bacteria)}=253$ ($P<0.001$); and for the 3rd
222 cycle $F_{Zn(matrix)}=69.4$ ($P<0.001$), $F_{Zn(bacteria)}=175$ ($P<0.001$) and $F_{Zn(matrix*bacteria)}=58.5$
223 ($P<0.001$).
224 For each specific matrix (alginate, pectate, synthetic polymer and incubated synthetic
225 polymer), the effect of the bacterial application on Zn removal was determined using
226 one way ANOVA. In the alginate matrix, generally inoculation with strain EC30
227 immobilised in alginate gave the best immobilisation of this metal (Table 2). The
228 removal varied significantly ($P<0.05$) within cycles of metal application, showing that a
229 clear relationship between the repeated use and the removal efficiency cannot generally
230 be drawn for alginate. For pectate-based treatments, generally strain 1ZP4 was the best

strain. However, in by the 3rd cycle there was no difference between treatments ($P<0.05$). Removals of Zn by the synthetic polymer matrix based treatments are also shown in Table 2. In general, strain 1C2 was more active when combined with the synthetic polymer. Over time (1-3 cycles) this combination became less efficient at removing this metal. When the bacterial cells were incubated with the synthetic polymer prior to packing, again strain 1C2 was the best treatment and t significantly ($P<0.05$) enhanced Zn removal in this matrix (Table 2). Overall, strain 1C2 immobilised on the synthetic polymer (PY+1C2) was the best treatment and was significantly ($P<0.05$) better (up to 76% more metal removed), than the other treatments especially in cycles 1 and 2. Effective removal was also observed for the polymer with EC30 (PY+EC30) and for both these combinations when bacteria were incubated with the polymer (PYInc+1C2 and PYInc+EC30).

Adsorption efficiencies to bacterial biomass per unit weight of cells were determined and are shown in Table 3 for each bacterial treatment. For Zn removal in single solutions, the best results were obtained for the PYInc+EC30, with an efficiency of 2.2 mg Zn/g bacterial cells.

Removal of Cd

The matrix type and bacterial strain immobilisation had a significant ($P<0.05$) effect on Zn removal (two-way ANOVA). In all cycles, the treatments that included bacteria showed significantly ($P<0.05$) better Cd removal than when the matrices were used alone. Test results were for the 1st cycle $F_{Cd(matrix)}=756$ ($P<0.001$), $F_{Cd(bacteria)}=1524$ ($P<0.001$) and $F_{Cd(matrix*bacteria)}=135$ ($P<0.001$); for the 2nd cycle $F_{Cd(matrix)}=185$ ($P<0.001$), $F_{Cd(bacteria)}=630$ ($P<0.001$) and $F_{Cd(matrix*bacteria)}=272$ ($P<0.001$); and for the 3rd cycle $F_{Cd(matrix)}=45.2$ ($P<0.001$), $F_{Cd(bacteria)}=645$ ($P<0.001$) and $F_{Cd(matrix*bacteria)}=209$ ($P<0.001$).

As for Zn, Cd removal was compared for each specific matrix treatment alone and with immobilised bacterial strains. Strain EC30 immobilised in alginate was shown to significantly immobilise this metal (Table 3). The behaviour of these combinations of alginate-bacteria was also analysed throughout the cycles and it generally varied with time, with significant ($P<0.05$) differences in the removal efficiencies between the 3 cycles. Strains 1ZP4 and 1C2 immobilised in pectate significantly ($P<0.05$) increased Cd removal. The behaviour of these pectate-bacteria combinations varied throughout

the cycles. Immobilisation with strain EC30 in the synthetic polymer gave a 11-fold increase in the removal of Cd when compared with the polymer alone; additionally, all the treatments showed a significant ($P<0.05$) decrease of removal efficiency of Cd throughout the cycles, similarly to what happened for Zn (Table 3) When the bacteria were incubated with the synthetic polymer prior to packing, no specific treatment was found to be more effective than any other. However, strains EC30 and 1C2 immobilised directly with the polymer matrix improved removal (Table 3). For all cycles, strain EC30 immobilisation onto the synthetic polymer (PY+EC30) was the best treatment.

Cadmium adsorption efficiencies per unit weight of cells (Table 3) in single solutions were determined and the best results were also obtained for the PYInc+EC30, with an efficiency of 2.8 mg Cd/g bacterial cells.

Removal of binary mixtures of metals by matrices and immobilised bacterial strains

The ability of the bacterial tested strains to take up metals from binary mixtures was then determined. Strain EC30 was best at removing Cd from the binary mixtures, regardless of immobilising system used (see Table 3). All the treatments showed significant ($P<0.05$) variations in the removal efficiencies of Cd throughout the cycles, according to one-way ANOVA performed on data. For Zn, strain EC30 immobilised in the alginate matrix improved the differential uptake ($P<0.05$) (Table 3), while strain 1ZP4 enhanced metal uptake when immobilised in pectate. Strain 1C2 was best at removing Zn from the binary mixtures when using the synthetic polymer. Overall, strain 1C2 + PY was best at differentially taking up Zn. As previously observed, by the 3rd cycle metal removal was much less than in the earlier cycles.

Zinc and Cd adsorption efficiencies per unit weight of cells in the binary solution were also determined (Table 3) and the best performance was of the treatments PYInc+1C2 and PY+1C2 for Zn, with an adsorption level of 1.8 mg Zn/g cells, and of P+1C2 and A+1C2 for Cd, registering efficiencies of 2.2 mg Cd/g cell.

Zn removal in single (Zn) and binary (Zn+Cd) mixtures in each treatment were also compared pair wise using the t-test (Table 2). For all matrices and cycles, differences in the ability to remove Zn were observed between simple and binary contamination scenarios, which seem to indicate that the performance of the treatments is influenced not only by the concentration but also by the metal feed composition. The same procedure was used for Cd removal in single (Cd) and binary (Zn+Cd) solutions (Table

3). As in the case of Zn, for all matrixes and cycles, differences in Cd removal were observed between simple and binary contamination scenarios.

Cd and Zn removal in the binary mixture were compared using the t-test. Results showing levels of the metals in the outlet (in mM) are presented in Figure 2 for alginate, and indicate that levels of Cd in the outlet were always significantly ($P<0.05$) lower than those of Zn. For pectate based combinations, the same trend was observed (Figure 3). With the exception of 1C2 immobilised to the synthetic polymer treatment, that presented no significant ($P<0.05$) differences in Cd and Zn removal in cycle 1 (Figure 4), levels of Cd at the outlet were significantly ($P<0.05$) lower than those of Zn in the polymer based treatments (Figures 4 and 5), decrease that showed to be of up to 65%. It seems thus that generally the tested bacteria-matrix combinations had higher affinity for Cd when a binary mixture was present.

Discussion

The aim of the work was to assess the effect of bacterial immobilisation in metal removal, and to compare the efficiency of bacteria + polymer combinations in order to understand which combinations were most appropriate for use in the clean-up of Cd and Zn contaminated waters.

Removal of individual metals by immobilised bacterial matrices

Metal sequestration by a sorbent may be due to one or a combination of the following processes: ion exchange, physical adsorption, chemisorptions, complexation or microprecipitation [32]). In the case of alginate − a linear polysaccharide that can be found in many algal species [33] and which has been extensively used in metal removal studies [34] − and pectate − a pectin compound which has been used to remove Zn in aqueous solutions by Khotimchenko et al. [13] − it appears that the process of ion-exchange takes place when metal binds to this matrix [35, 36].

Despite this adsorption capacity of the polymers, the present study showed that the immobilisation of bacteria increased the removal abilities of all the matrices (alginate, pectate and the synthetic polymer). In fact, bacteria have been successfully used as biosorbents [7, 8, 9] because of their small size, their ubiquity, ability to grow under controlled conditions and resilience to a wide range of contaminants [37]. Bacteria are known to produce extracellular polymeric substances which are composed by proteins,

polysaccharides and uronic acid. These substances contain several functional groups like carboxyl, phosphoric, amine and hydroxyl groups [38, 39]. Both the phosphoryl and carboxyl groups of the peptide chains in bacterial cell walls provide negatively charged sites in Gram-positive bacteria. For Gram-negative bacteria, such as 1ZP4, EC30 and 1C2, the phosphate groups within the lipopolysaccharides of their outer membrane are the primary sites for metal interaction, with only one of the carboxyl group in this net being free to interact with metals [37]. The process of binding of metal ions to bacteria involves electrostatic interaction between metal ions and the biomass [4] as bacteria have a net negative charge that favour the biosorption of metal [40], as observed in the present work. Further studies have shown a similar pattern when comparing the use of polymers alone and when immobilizing microorganisms: For example, Sag *et al*. [41] have shown that when aqueous solutions of Cu were treated with Ca-alginate immobilised *Zooglea ramigera*, an increase in Cu removal occurred from 64 %, for the treatment with only Ca-alginate, to 94 %. Aksu et al. [11] have also shown that after long periods, the adsorption capacity of alginate immobilised *Chlorella vulgaris* exceeded that of alginate alone. Synthetic responsive polymers have also been used successfully to control the attachment of bacterial cells to surfaces [42] demonstrating the attachment of *Hallomonas* and *Staphylococcus* strains to surface-grafted synthetic polymers. However, the amount of biosorbent, initial concentration of metal, presence of further contaminants in the aqueous solutions, structural properties of both the support matrix and the biosorbent material all affect the biosorption rate [34], rendering it difficult to compare results from different reports, and thus the main focus of this report is not to attempt such comparisons. The 3 selected strains – 1C2, 1ZP4 and EC30 – exhibited high resistance to Cd and Zn and all showed high specific growth rates when these heavy metals were present at different concentrations. Strains 1C2, 1ZP4 and EC30 are all Gram-negative and affiliated to genera *Cupriavidus*, *Sphingobacterium* and *Alcaligenes*, respectively. Many reports have shown that Gram-negative are more tolerant to heavy metals than Gram-positive bacteria. This metal tolerance can be attributed to the interactions between microbial cell wall components and heavy metal ions both contributing to metal detoxification [43, 44, 45]. In the biosorption of complex solutions, different metal ions may compete for the active sites existing on the support matrix and/or on the cell wall of the biomass. Consequently, the preference of the biomass for some metals is an important issue [46], and thus the

363 knowledge of the growth and metal resistance patterns of the bacterial species is of great
364 importance.
365 Measurement of the growth of the selected strains in the presence of Cd and Zn
366 indicated differences in toxicity towards the bacteria among the heavy metals.
367 Specifically, the presence of Cd^{2+} inhibited the growth of the strains tested, except for
368 strain EC30 that showed a remarkable capacity to tolerate Cd in solution, with only a
369 15-20 % biomass reduction. Zn^{2+} caused also a reduction in biomass production;
370 however in a less significant degree when compared to Cd. Strain EC30 apparently was
371 more sensitive to Zn^{2+} than to Cd^{2+}. When metal mixtures were present, the growth rate
372 was lower than that observed when only Zn was tested. The decrease in biomass
373 observed whenever metals were present possibly results from a decrease in the substrate
374 utilization efficiency due to a higher energy cost of microorganisms subject to metal
375 stress [47].
376 In the present study 1C2, a species affiliated to the *Cupriavidus* genera, was generally
377 the one that most increased the removal performance of Zn (in single and binary
378 solutions), especially when associated with the synthetic polymer. In contrast, EC30, a
379 bacterium affiliated to the *Alcaligenes* genera, gave the most promising results for Cd
380 removal in single and binary mixtures, especially when combined with the synthetic
381 polymer. In fact, EC30 has also shown to be the most resistant to Cd in the tolerance
382 study performed which may explain these results. Mondal et al. [48] reported the use a
383 species of *Ralstonia*, phylogenetically related to *Cupriavidus*, *Ralstonia eutropha*, for
384 the elimination of Fe, Mn, Cu, As and Zn, with removals of up to 65.2, 72.7, 98.6, 8 %
385 and 99.3 % respectively from metal contaminated water. Species from the genera
386 *Alcaligenes* (such as EC30) have also been reported by Chang and Tseng [49] as
387 important in immobilised biomass strategies, and Diels et al. [50] have studied the
388 application for heavy metal removal of composite membrane reactor immobilised
389 *Alcaligenes eutrophus* bacteria with a reduction of metals such as Cd, Zn, Cu, and Pb in
390 solution from 100 ppm to less than 50 ppb. As for strain 1ZP4, belonging to genera
391 *Sphingobacterium*, there is also a study from Bootham et al. [51] describing
392 *Sphingobacterium mizutatae* as being part of a bacterial consortium used to treat metal
393 contaminated effluents.
394 The removal efficiencies registered in the present report reach maximum levels of 2.8
395 mg Cd/ g cell and 2.2 mg Zn/g cell. Yakup Arica et al. [34] used Ca alginate as a

support for Zn biosorption with immobilized live and inactivated fungus *Phanerochaete chrysosporium*, and for a similar initial Zn concentration (100 mg l^{-1}) removals of ca. 20 to 35 mg Zn g^{-1} adsorbent were observed. In fact, these values are quite higher than the ones shown in our study, however the residence time was of 90 min while in our study average contact times of 2 min were used. Also, and for solution of similar Cd initial concentration, Quintelas et al. [2] presented uptake levels of app. 10 mg Cd g^{-1} *Escherichia coli* supported on kaolin, this time for a residence time of 10 days. Nevertheless, the levels of adsorption of the tested systems will depend not only on the characteristics of the used immobilization media, but also on the residence time of the metals in the cartridge. Sag et al. [41] analysed the effect of flow rate in the adsorption of Cu to alginate and immobilized *Zooglea ramigera* and have showed that an increase in the flow of 5 times could result in decreases in the metal removal of up to 15 times.

Removal of binary mixtures of metals by immobilised bacterial matrices

Mixtures of Cd and Zn are typically found in contaminated effluents of industrial processes [52]; additionally, from a biological point of view Cd can be transported by the same transporters as Zn [53].Nevertheless, Fan et al. [54] have shown that when using binary mixtures of Cd and Zn, the biosportion capacity of either metal was lower than that found in non-competitive conditions. However, this did not always occur in the present study. In some cases there was a differential increase in the removal abilities of either of the tested metals when present as a binary solution when compared to single solution. Such phenomenon may be explained by the hypothesis that the sorption of the other metallic contaminants in solution altered the conformation of the metal binding sites and increased the affinity of sites for that particular metal adsorption in that specific combination of matrix, bacteria and usage [10]. On the other hand, the opposite effect was observed in some cases where there was a decrease in Cd or Zn removal capacities of specific matrix-bacteria combinations. The most likely reason for this antagonistic effect may be the competition for adsorption sites on the cell and polymer surfaces. Chen et al. [10] also found that Cd uptake capacity was slightly reduced when Pb and Hg are present in solution, suggesting that in Ca-alginate immobilised *Microcystis aeruginosa* most Cd adsorption sites were specific, whereas some of these Cd binding sites were also capable of binding other metals. Despite these variations in the removal of metals in the binary mixture levels of Cd at the outlet were lower than

429 those of Zn, and in the large majority of cases this trend was significant. The preference
430 of a sorbent for a metal may be explained on the basis of electronegativity of the metal
431 ions (Cd=1,69 and Zn= 1,65, according to the Pauling scale), molecular weight
432 (Cd=112,4 and Zn=65,4) and ionic radius (Cd=95 and Zn=74), with the first being
433 positively related to the adsorption capacity, and the second and third being inversely
434 related to it [2]. In the present study, electronegativity seems to play an important role in
435 the affinity of the tested combinations to Cd, but other conditions such as ionization
436 energy can have contributed to influence the adsorption behavior of the metals [55].
437

438 **Conclusions**

439 Immobilisation of bacteria in naturally occurring and synthetic polymers increased the
440 removal abilities of all the matrixes (alginate, pectate and synthetic cross-linked
441 polymer), with up to 12-fold when compared to the use of the polymers alone. Strain
442 1C2, a species from the *Cupriavidus* genera, generally has the best capacity for
443 increasing the removal of Zn when immobilised on any of the polymers, in single and
444 binary solutions, especially when associated with the synthetic polymer. EC30, a
445 bacteria affiliated to the *Alcaligenes* genera, was the most promising concerning Cd
446 removal in single and binary mixtures, again when combined with the synthetic
447 polymer. Thus, the combinations that would be recommended to clean-up aqueous
448 solutions containing Zn or Cd would be respectively 1C2 or EC30 immobilised on the
449 synthetic polymer (PY+1C2 and PY+EC30). Synthetic cross-linked polymers are
450 promising matrixes and should be explored further in immobilised microbial cartridges. In
451 this format, in addition to the promising results presented here, synthetic polymers have the
452 added advantage of being easily reusable, unlike their natural counterparts.
453

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460 **References**

461 [1] J. Wang, C. Chen, Biosorbents for heavy metals removal and their future,
462 Biotechnol. Adv., 27 (2009) 195-226.
463 [2] C. Quintelas, Z. Rocha, B. Silva, B. Fonseca, H. Figueiredo, T. Tavares, Removal of
464 Cd(II9, Cr(VI), Fe(III) and Ni(II) from aqueous solutions by an *E. coli* biofim
465 supported on kaolin, Chem. Eng. J., 149 (2009) 319-324.
466 [3] B. Volesky, Detoxification of metal bearing effluents: biosorption for the next
467 century, Hydrometallurgy, 59 (2001) 203-216.
468 [4] K.K. Krishnani, X. Mengb, C. Christodoulatos, V.M. Bodduc, Biosorption
469 mechanism of nine different heavy metals onto biomatrix from rice husk, J. Haz.
470 Mat., 153 (2008) 1222-1234.
471 [5] E.I. Unuabonah, K.O Adebowale, B.I. Olu-Owolabi, Kinetic and thermodynamic
472 studies of the adsorption of lead (II) ions onto phosphate modified kaolinite clay, J.
473 Haz. Mat., 144 (2007) 386-395.
474 [6] A.I. Zouboulis, M.X. Loukidou, K.A. Matis, Biosorption of toxic metals from
475 aqueous solutions by bacteria strains isolated from metal polluted soils, Process
476 Biochem., 39 (2004) 909-916.
477 [7] A. Nakajima, T. Tsaruta, Competitive biosorption of thorium and uranium by
478 *Micrococcus luteus*, J. Radioanal. Nuc. Chem., 260 (2004) 13-18.
479 [8] A. Selatnia, M.Z. Bakhti, A. Madani, I. Kertous, Y. Mansouri, Biosorption of Cd2+
480 from aqueous solution by a NaOH treated bacterial dead *Streptomyces rimosus*
481 biomass, Hydrometallurgy, 75 (2004) 11-24.
482 [9] M. Ziagova, G. Dimitriadis, D. Aslanidou, X. Papaioannou, E.I. Tzannetaki, M.
483 Liakopoulou-Kyriakides, Comparative study of Cd(II) and Cr(VI) biosorption on
484 *Staphylococcus xylosus* and *Pseudomonas* sp. in single and bynary mixtures, Biores.
485 Technol, 98 (2007) 2859-2865.
486 [10] J.Z. Chen, X.C. Tao., T. Zhang., Z.L. Liu, Biosorption of lead, cadmium and
487 mercury by immobilised *Microcystis aeruginosa* in a column. Process Biochem, 40
488 (2005) 3675-3679.
489 [11] Z. Aksu, G. Egretli, T. Kutsal, A comparative study of copper (II) biosorption on
490 Ca-alginate, agarose and immobilised *C. vulgaris* in a packed bed colum, Process
491 Biochem., 33 (1998) 393-400.
492 [12] K. Akhtar, M.W. Akhtar, A.M. Khalid, Removal and recovery of uranium from
493 aqueous solutions by *Trichoderma harzianum*, Water Res., 41 (2007) 1366-1378.

494 [13] M.Y. Khotimchenko, E.A. Kolenchemko, Y.S. Khotimchenko, Zinc binding
495 activity of different pectin compounds in aqueous solutions, J. Colloid Interface Sci.,
496 323 (2008) 216-222.
497 [14] C. de las Heras Alarcón, B. Twaites, D. Cunliffe, J.R. Smith, C. Alexander, Grafted
498 thermo- and pH responsive co-polymers: Surface-properties and bacterial adsorption,
499 Int. J. Pharmaceut. 295(2005) 77–91.
500 [15] C.F. Degiorgi, R.A. Pizarro, E.E. Smolko, S. Lora, M. Carenza, Hydrogels for
501 immobilisation of bacteria used in the treatment of metal contaminated wastes,
502 Radiation Physics Chem., 63 (2002) 109-113.
503 [16] S. Mor, K. Ravindra, N.R. Bishnoi, Adsorption of chromium from aqueous
504 solution by activated alumina and activated charcoal, Biores. Technol., 98 (2007)
505 954-957.
506 [17] V.P.J. Vilar, R.J.E. Martins, C.M.S. Botelho, R.A.R. Boaventura, Removal of Cu
507 and Cr from an industrial effluent using a packed bed column with algae Gelidium-
508 derived material, Hydrometallurgy, 96 (2009) 42-46.
509 [18] A.P.G.C. Marques, A.O.S.S. Rangel, P.M.L. Castro, Zn accumulation in plant
510 species indigenous to a Portuguese polluted site: relation with soil contamination, J.
511 Environ. Qual. 36 (2007) 646-653.
512 [19] I. Tiago, I. Teixeira, S. Silva, P. Chung, A. Veríssimo, C. Manaia, Metabolic and
513 genetic diversity of mesophilic and thermophilic bacteria isolated from composted
514 municipal sludge on poly-epsilon-caprolactones, Cur. Microbiol., 49 (2004) 407-
515 414.
516 [20] S.K. Alexander, D. Strete, Microbiology: a Photographic Atlas for the Laboratory,
517 2nd edn. San Francisco: Benjamin-Cummings Publishing Company (2001).
518 [21] R.G.E. Murray, R.N. Doetsch, F. Robinow,. Determinative and cytological. light
519 microscopy. In Methods for General and Molecular Bacteriology, Edited by P.
520 Gerhardt, R. G. E. Murray, W. A. Wood & N. R. Krieg. Washington, DC: American
521 Society for Microbiology (1994) pp. 21–41.
522 [22] R.M. Smibert, N.R. Krieg, Phenotypic characterization. In Methods for General
523 and Molecular Bacteriology, Edited by P. Gerhardt, R. G. E. Murray, W. A. Wood
524 & N. R. Krieg. Washington, DC: American Society for Microbiology (1994), pp.
525 611–651.
526 [23] G. Gomori, Preparation of buffers. Methods in Enzymology, 1 (1990) 138–146.

[24] F.A. Rainey, N. Ward-Rainey, R.M. Kroppenstedt, E. Stackebrandt, The genus *Nocardiopsis* represents a phylogenetically coherent taxon and a distinct actinomycete lineage: proposal of Nocardiopsaceae fam. Nov, Int. J. Systemat. Bacteriol., 46 (1996) 1088–1092.

[25] D.J. Lane, 16S/23S sequencing. In Nucleic Acid Techniques in Bacterial Systematics,. Edited by E. Stackebrandt & M. Goodfellow. Chichester: Wiley (1991), pp. 171–204.

[26] T.A. Hall, BioEdit: a user-friendly biological sequence alignment editor and analysis program for Windows 95/98/NT. Nucleic Acids Symp Ser 41 (1999) 95–98.

[27] D.A. Benson, I. Karsch-Mizrachi, D.J. Lipman, J. Ostell, D.L. Wheeler, GenBank. Nucleic Acids Res., 35 (2007) 21–25.

[28] N.W. Turner, A.P.F. Turner, S.A. Piletsky, Controlled release of the herbicide simazine from computationally designed molecularly imprinted polymers, J. Controlled Release, 108 (2005) 132-139.

[29] **S. Barral, A. Guerreiro, M. Villa-García, M. Rendueles, M. Díaz, S. Piletsky,** Synthesis of 2-(diethylamino) ethyl methacrylate-based polymers: Effect of crosslinking degree, porogen and solvent on the textural properties and protein adsorption performance, Reactive and functional polymers, 70 (2010) 890-899.

[30] S.E.M. Escamilla, L. Dendooven, I.P. Magaña, S.R. Parra, M.M. De la Torre, Optimization of gibberellic acid production by immobilised *Gibberella fujikuroi* mycelium in fluidized bioreactors, J. Biotechnol, 76 (2000) 147-155.

[31] M.C. Montes, I.P. Magana, Delta'-dehydrogenation of steroids by *Arthrobacter simplex* immobilised in calcium polygalacturonate beads, J. Industrial Microbiol, 8 (1991) 259-264.

[32] V. Utgikar, B.Y. Chen, H.H. Tabak, D.F. Bishop, R. Govnd, Treatent of acid mine drainage. Equilibrium biosorption of zinc and copper on non viable activated sludge, Int. Biodeter. Biodegrad, 46 (2000) 19-28.

[33] B. Stokke, O. Smidsrod, P. Bruheim, G. Skjak-Braek, Distribution of urinate residues in alginate chains in relation to alginate gelling properties, Macromolecules, 24 (1991) 4637-4645.

559 [34] M.Y. Arica, C. Arpa, A. Ergene, G. Bayramoglu, O. Genç, Ca-alginate as a support
560 for Pb(II) and Zn(II) biosorption with immobilised *Phanerochaete chrysosporum*,
561 Carbohydrate Polym., 52 (2003) 167-174.
562 [35] S. Myklestad, Ion exchange properties of brown algae. Determination of rate
563 mechanism for calcium hydrogen ion exchange for particles from *Laminaria*
564 *hyperborean* and *Laminaria digitata*, J. Applied Chem., 18 (1968) 30-36.
565 [36] A.J. Paskins-Hurlburt, Y. Tanaka, S.C. Skoryna, W. Moore, J.F. Stara The binding
566 of lead by pectic polyelectrolyte, Environment. Res., 14 (1977) 128-140.
567 [37] M.M. Urrutia, General Bacterial Sorption Processes. In: Wase, J., Forster, C. (eds)
568 Biosorbents for metal ions. London, UK, CRC Press, (1997) p.39-66.
569 [38] E.D. Van Hullebush, M.H. Zandvoort, P.N.L. Lens, Metal immobilisation by
570 biofilms: mechanisms and analytical tools, Rev. Environment. Sci. Biol. Technol., 2
571 (2003) 9-33.
572 [39] A. Omoike, J. Chorover, Spectroscopic study of extracellular polymeric substances
573 from *Bacillus subtilis*: aqueous chemistry and adsorption effects,
574 Biomacromolecules, 5 (2004) 1219-1230.
575 [40] B. Silva, H. Figueiredo, C. Quintelas, T. Tavares, I.C. Neves, Zeolites as supports
576 for the biorecovery of hexavalent and trivalent chromium, Microporous Mesoporous
577 Mat., 116 (2008) 555-560.
578 [41] Y. Sag, Z. Nourbakhsh, Z. Aksu, T. Kutsal, Comparison of Ca-alginate and
579 immobilised *Z. ramigera* as sorbents for copper removal, Process Biochem., 30
580 (1995) 175-181.
581 [42] L.K. Ista, V.H. Perez-Luna, G.P. Lopez, Surface-Grafted, Environmentally
582 Sensitive Polymers for Biofilm Release, Applied Environment. Microbiol., 65
583 (1999) 1603-1609.
584 [43] Å. Frostegård, A. Tunlid, E. Bååth, Phospholipid fatty acid composition, biomass,
585 and activity of microbial communities from two soil types experimentally exposed
586 to different heavy metals, Applied Environment. Microbiol., 59 (1993) 3605-3617.
587 [44] T.M. Roane, S.T. Kellog, Characterisation of bacterial communities in heavy metal
588 contaminates soils, Canad. J. Microbiol., 42 (1996) 593-603.
589 [45] T. Pennanen, A. Frostegård, H. Fritz, Phospholipid fatty acid composition and
590 heavy metal tolerance of soil microbial communities along two heavy metal-polluted
591 gradients in coniferous forest, App. Environment. Microbiol., 62 (1996) 420-428.

592 [46] E. Romera, F. Gonzalez, A. Ballester, M.L. Blazquez, J.A. Munoz, Biosorption of
593 heavy metals by *Fucus spiralis*, Biores. Technol, 99 (2008) 4684-4693.
594 [47] K.E. Giller, E. Witter, S.P. McGrath, Heavy metals and soil microbes, Soil Biol.
595 Biochem., 41 (2009) 2031-2037.
596 [48] P. Mondal, C.B. Majumber, B. Mohanty, Treatment of arsenic contaminated water
597 in a batch reactor by using *Ralstonia eutropha* MTCC 2487 and granular activated
598 carbon, J. Haz. Mat., 153 (2008) 588-599.
599 [49] C.C. Chang, S.K. Tseng, Immobilisation of *Alcaligenes eutrophus* using PVA
600 crosslinked with sodium nitrate, Biotechnology. Tech., 12 (1998) 865–868.
601 [50] L. Diels, S. van Roya, K. Somersa, I. Willemsa, W. Doyenb, M. Mergeaya, D.
602 Springaela, R. Leysenb, The use of bacteria immobilised in tubular membrane
603 reactors for heavy metal recovery and degradation of chlorinated aromatics, J.
604 Membrane Sci., 100 (1995) 249-258.
605 [51] C. Boothman, S. Hockin, D.E. Holmes, G.M. Gadd, J.R. Lloyd, Molecular analysis
606 of a sulphate-reducing consortium used to treat metal-containing effluents,
607 BioMetals, 19 (2006) 601–609.
608 [52] Baird, C. & Cann, M. Environmental Chemistry. 3rd Ed.. New York: *W.H.*
609 *Freeman and Company*. (2005).
610 [53] McEldowney, S., (1990). Microbial biosorption of radionuclides in liquid effluent
611 treatment. Appl. Biochem. Biotechnol. 26 (1990) 159–180.
612 [54] T. Fan, Y. Liu, B. Feng, G. Zeng, C. Yang, M. Zhou, H.Zhou, Z. Tan, X. Wang,
613 Biosorption of cadmium (II), zinc (II) and lead (II9 by *Penicillium simplicissimum*:
614 isotherms, kinetics and thermodynamics, J. Haz. Mat., 160 (2008) 655-661.
615 [55] A.S. Ekop, N.O. Eddy, Adsorption of Pb2+, Zn2+ and Ni2+ from Aqueous
616 Solution by *Helix asper*a shell. E-Journal Chem., 6 (2009) 1029-1034.
617

Table 1: Levels of Zn in the outlet for each treatment (mg Zn/L)

Treatment	Round 1		Round 2		Round 3	
	100 mg Zn/L	100 mgZn + 100 mgCd / L	100 mg Zn/L	100 mgZn + 100 mgCd / L	100 mg Zn/L	100 mgZn + 100 mgCd / L
A	97.4 ± 0.1 ^{gh,D}	92 ± 2 $^{ef,C\neq}$	84 ± 2 ^{ef,B}	84 ± 6 ^{fg,BC}	82 ± 1 ^{ef,B}	76 ± 0 ^{abc,AB}
A + 1C2	83.4 ± 0.5 ^{ef,C}	81 ± 2 ^{de,B}	87.8 ± 0.3 ^{f,B}	89.8 ± 0.8 $^{g,C\neq}$	87,1 ± 0.6 ^{f,C}	83 ± 2 ^{abc,C}
A + 1ZP4	52 ± 3 ^{c,A}	66 ± 5 $^{c,A\neq}$	84 ± 2 ^{efv,B}	75.7 ± 0.09 $^{df,AB\neq}$	79 ± 4 ^{def,B}	79 ±2 ^{abc,BC}
A + EC30	64.9 ± 0.2 ^{d,B}	70 ± 2 $^{cd,A\neq}$	69 ± 2 ^{d,A}	74 ± 3 ^{d,A}	71.0 ± 0.8 ^{cd,A}	73.9 ± 0.8 $^{abc,A\neq}$
	****F=513*	****F=44.2*	****F= 67.1*	***F=14,5*	****F=35.6*	***F=16.3*
P	91 ± 1 ^{gh,C}	99 ± 2 $^{f,C\neq}$	79.4 ± 0.7 ^{e,B}	77 ± 4 ^{df,A}	74 ± 2 ^{cde,A}	65 ± 0 $^{a,A\neq}$
P + 1C2	79 ± 2 ^{e,B}	77.8 ± 0.4 ^{d,B}	80 ± 3 ^{e,B}	83.5 ± 0.6 ^{fg,B}	79 ± 6 ^{def,A}	82.425 ± 0.005 ^{abc,D}
P + 1ZP4	41 ± 2 ^{b,A}	44 ± 2 ^{b,A}	68 ± 3 ^{d,A}	74 ± 2 $^{cd,A\neq}$	79.9 ± 0.8 ^{defg,A}	75.5 ± 0.6 $^{abc,B\neq}$
P + EC30	80.28 ± 0.03 ^{e,B}	74 ± 3 $^{cd,B\neq}$	80 ± 1 ^{e,B}	76.8 ± 0.3 $^{df,A\neq}$	77 ± 2 ^{de,A}	79.7 ± 0.5 $^{abc,C\neq}$
	****F=588*	****F=386*	****F=21.0*	***F=10.4*	^{NS}*F=2.14*	**** F=734*
PY	102.05 ± 0.05 ^{h,C}	101.4 ± 0.6 ^{f,C}	105.5 ± 0.8 ^{h,D}	106.2 ± 0.6 $^{h,D\neq}$	109.0 ± 0.4 ^{g,B}	108.6 ± 0.3 ^{c,A}
PY + 1C2	26 ± 4 ^{a,A}	22 ± 6 ^{a,A}	31.9 ± 0.5 ^{a,A}	42,0 ± 0.1 $^{a,A\neq}$	74 ± 8 ^{cde,A}	72.3 ± 0.2 ^{abc,A}
PY + 1ZP4	44 ± 4 ^{b,B}	46 ± 2 ^{b,B}	68 ± 2 ^{d,C}	73 ± 2 ^{cd,C}	99 ± 1 ^{g,B}	101 ± 1 ^{abc,A}
PY + EC30	22 ± 2 ^{a,A}	35 ± 12 ^{b,AB}	50 ±2 ^{c,B}	64 ± 5 ^{bc,B*}	65 ± 4 ^{c,A}	76 ± 6 ^{abc,A}
	****F=477*	****F=82.6*	****F=1118*	****F=305*	****F=64.6*	^{NS}*F=1.41*
PYInc	96 ± 4 ^{gh,C}	101 ± 1 ^{f,C}	96 ± 1 ^{g,C}	103.99 ± 0.06 ^{h,C}	101 ± 2 ^{g,D}	106 ± 4 ^{bc,C}
PYInc + 1C2	28 ± 4 ^{a,AB}	18.9 ± 0.5 $^{a,A\neq}$	41 ± 4 ^{b,A}	48 ± 7 ^{a,A}	51 ± 3 ^{b,B}	67 ± 2 $^{ab,A\neq}$
PYInc + 1ZP4	37 ± 4 ^{b,B}	44 ± 3 ^{b,B}	47.5 ± 0.4 ^{c,B}	44 ± 4 ^{a,A}	79 ± 3 ^{def,C}	79 ± 3 ^{abc,B}
PYInc + EC30	25 ± 3 ^{a,A}	21 ± 1 ^{a,A}	38.7 ± 0.4 ^{b,A}	60 ± 2 $^{b,B\neq}$	39 ± 4 ^{a,A}	70 ± 2 ^{abc,A}
	****F=277*	****F=1503*	****F=520*	****F=140*	****F=254*	****F=118*
	*** (F=404)	*** (F=172)	*** (F=387)	*** (F=108)	*** (F=84)	* (F=2.52)

Results are expressed as mean ± S.D. (n = 3). Means for each treatment in the same column with different lowercase letters are significantly different from each other ($P < 0.05$) according to the Tukey test. For each round, the test results are shown with the test statistics and as: NS, non-significant at the level $P < 0.05$; *significant at the level $P < 0.05$; **significant at the level $P < 0.01$; ***significant at the level $P < 0.001$.
For each matrix (alginate, pectate, polymer and incubated polymer) results of one way ANOVA are also shown with the test statistics and as: NS, non-significant at the level $P < 0.05$; *significant at the level $P < 0.05$; **significant at the level $P < 0.01$; ***significant at the level $P < 0.001$. Means for the same matrix type in the same round with different uppercase letters are significantly different from each other ($P < 0.05$) according to the Tukey test.
Results of the comparison between results for different effluents (Zn and Zn+Cd) for each treatment are shown and when means of Cd+Zn in each round have a ≠ signal they are significantly different from means of outlet Zn ($P < 0.05$) according to the t-test.

Table 1: Levels of Cd in the outlet for each treatment (mg Cd/L)

Treatment	Round 1		Round 2		Round 3	
	100 mg Cd/L	100 mgZn + 100 mgCd / L	100 mg Cd/L	100 mgZn + 100 mgCd / L	100 mg Cd/L	100 mgZn + 100 mgCd / L
A	88 ± 2 ^{f,C}	85 ± 2 ^{h,A}	61.5 ± 0.3 ^{f,B}	61 ± 2 ^{def,A}	63.1 ± 0.1 ^{e,B}	60 ±1 $^{a,AB\neq}$
A + 1C2	67.9 ± 0.2 ^{e,B}	72 ± 1 ^{gB}	68 ± 2 ^{fg,C}	65 ± 2 ^{ef,A}	69 ± 1 ^{e,B}	68.0 ± 0.9 ^{bcde,C}
A + 1ZP4	63.3 ± 0.4 ^{de,B}	58.47 ± 0.05 $^{d,A\neq}$	64 ± 1 ^{fg,BC}	61 ± 2 ^{def,A}	66 ± 5 ^{e,B}	58 ± 3 ^{a,A}
A + EC30	47 ± 4 ^{c,A}	60 ± 1 $^{de,A\neq}$	45 ± 2 ^{d,A}	63 ± 1 $^{ef,A\neq}$	48 ± 1 ^{cd,A}	62.7 ± 0.4 $^{abc,B\neq}$
	****F=147*	****F=239*	****F=144*	^{NS} *F=3.18*	****F=35.6*	****F=22.4*
P	92 ±3 ^{f,C}	86 ± 1 $^{h,B\neq}$	54 ± 2 ^{e,A}	58 ± 3 ^{de,A}	65.7 ± 0.3 ^{e,A}	64 ± 7 ^{abcd,AB}
P + 1C2	63 ± 1 ^{de,AB}	64.1 ± 0.4 ^{ef,A}	69.2 ± 0.7 ^{g,B}	69 ± 1 ^{f,C}	65 ± 1 ^{e,A}	69.6 ± 0.2 $^{de,B\neq}$
P + 1ZP4	58 ± 3 ^{d,A}	61.9 ± 0.8 ^{def,A}	64.8 ± 0.8 ^{fg,B}	64.9 ± 0.5 ^{ef,B}	65 ± 2 ^{e,A}	57 ± 3 $^{a,A\neq}$
P + EC30	68 ± 4 ^{e,B}	64 ± 3 ^{f,A}	64 ± 5 ^{fg,B}	59.92 ± 0.07 ^{def,A}	64 ± 2 ^{e,A}	61 ± 1 ^{a,AB}
	****F=87.7*	****F=76.0*	****F=18.7*	****F=34.8*	^{NS} *F=1.03*	**F=6.32*
PY	91.9 ± 0.3 ^{f,C}	98.24 ± 0.03 $^{i,C\neq}$	92.46 ± 0.07 ^{h,D}	96.9 ± 0.2 $^{g,C\neq}$	95.7 ± 0.5 ^{f,C}	100 ± 2 $^{f,C\neq}$
PY + 1C2	21 ± 1 ^{b,B}	40 ± 1 $^{c,B\neq}$	36 ± 3 ^{c,}	63.8 ± 0.9 $^{ef,B\neq}$	49 ± 3 ^{cd,B}	69 ± 2 $^{cdeB,\neq}$
PY + 1ZP4	6 ± 2 ^{a,A}	38.41 ± 0.05 $^{c,B\neq}$	25 ± 4 ^{bc,B}	33 ± 4 ^{a,A}	46 ± 4 ^{b,B}	61.7 ± 0.7 $^{ab,A\neq}$
PY + EC30	5 ± 1 ^{a,A}	23 ± 1 $^{b,A\neq}$	15.8 ± 0.7 ^{a,A}	38 ± 6 $^{a,A\neq}$	31 ± 1 ^{a,A}	58 ± 3 $^{a,A\neq}$
	****F=3860*	****F=5269*	****F=680*	****F=431*	****F=353*	****F=295*
PYInc	101.65 ± 0.05 ^{g,C}	101.25 ± 0.05 $^{i,C\neq}$	107.6 ± 0.2 ^{i,C}	106 ± 3 ^{g,C}	105.3 ± 0.7 ^{g,D}	105 ± 1 ^{f,D}
PYInc + 1C2	18 ± 3 ^{b,AB}	18.5 ± 0.5 ^{ab,A}	25 ± 3 ^{b,A}	46 ± 7 $^{bc,AB\neq}$	30 ± 2 ^{a,A}	69 ± 1 $^{cde,B\neq}$
PYInc + 1ZP4	19 ± 5 ^{b,B}	37 ± 3 $^{c,B\neq}$	37 ± 5 ^{c,B}	37 ± 3 ^{ab,A}	54.6 ± 0.8 ^{d,C}	72.2 ± 0.7 $^{e,C\neq}$
PYInc + EC30	11 ± 3 ^{a,A}	16.4 ± 0.5 $^{a,A\neq}$	22 ± 2 ^{ab,A}	52 ± 7 $^{cd,B\neq}$	41 ± 2 ^{b,B}	61.6 ± 0.6 $^{ab,A\neq}$
	****F=528*	****F=2052*	****F=476*	****F=96.3*	****F=1607*	****F=1112*
	*** (F=537)	*** (F=914)	*** (F=326)	*** (F=109)	*** (F=263)	*** (F=118)

Results are expressed as mean ± S.D. (n = 3). Means for each treatment in the same column with different lowercase letters are significantly different from each other ($P < 0.05$) according to the Tukey test. For each round, the test results are shown with the test statistics and as: NS, non-significant at the level $P < 0.05$; *significant at the level $P < 0.05$; **significant at the level $P < 0.01$; ***significant at the level $P < 0.001$.
For each matrix (alginate, pectate, polymer and incubated polymer) results of one way ANOVA are also shown with the test statistics and as: NS, non-significant at the level $P < 0.05$; *significant at the level $P < 0.05$; **significant at the level $P < 0.01$; ***significant at the level $P < 0.001$. Means for the same matrix type in the same round with different uppercase letters are significantly different from each other ($P < 0.05$) according to the Tukey test.
Results of the comparison between results for different effluents (Cd and Zn+Cd) for each treatment are shown and when means of Cd+Zn in each round have a ≠ signal they are significantly different from means of outlet Cd ($P < 0.05$) according to the t-test.

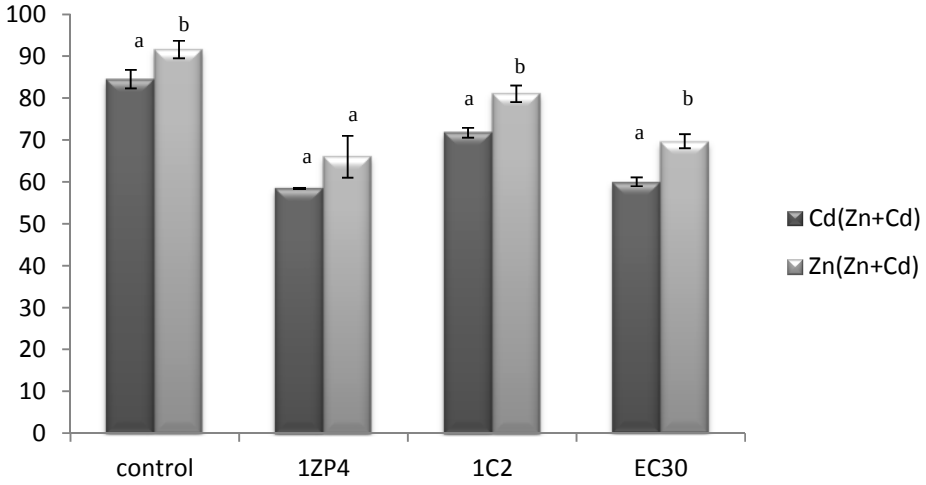
Table 3: Adsorption of metal per unit weight of cells for each treatment (mg Zn/g cell)

Treatment	Zn		Cd	
	100 mg Zn/L	100 mgZn + 100 mgCd / L	100 mg Cd/L	100 mgZn + 100 mgCd / L
A + 1C2	0.9 ± 0.1	1.0 ± 0.3	2.15 ± 0.09	2.2 ± 0.2
A + 1ZP4	1.3 ± 0.7	1.2 ± 0.3	1.6 ± 0.1	1.89 ± 0.09
A + EC30	1.2 ± 0.1	1.1 ± 0.1	2.1 ± 0.1	1.49 ± 0.06
P + 1C2	1.4 ± 0.2	1.3 ± 0.2	2.3 ± 0.2	2.2 ± 0.2
P + 1ZP4	1.7 ± 0.8	1.7 ± 0.7	1.7 ± 0.2	1.8 ± 0.2
P + EC30	0.82 ± 0.08	0.9 ± 0.1	1.4 ± 0.1	1.5 ± 0.1
PY + 1C2	1.9 ± 0.8	1.8 ± 0.7	2.2 ± 0.4	1.4 ± 0.4
PY + 1ZP4	1.0 ± 0.8	1.2 ± 1.0	2.5 ± 0.6	1.9 ± 0.4
PY + EC30	1.8 ± 0.6	1.4 ± 0.6	2.8 ± 0.4	2.0 ± 0.5
PYInc + 1C2	2.0 0.3	1.8 ± 0.7	2.5 ± 0.2	1.9 ± 0.7
PYInc + 1ZP4	1.5 ± 0.6	1.5 ± 0.6	2.1 ± 0.5	1.7 ± 0.6
PYInc + EC30	2.2 ± 0.2	1.7 ± 0.8	2.5 ± 0.5	1.9 ± 0.7

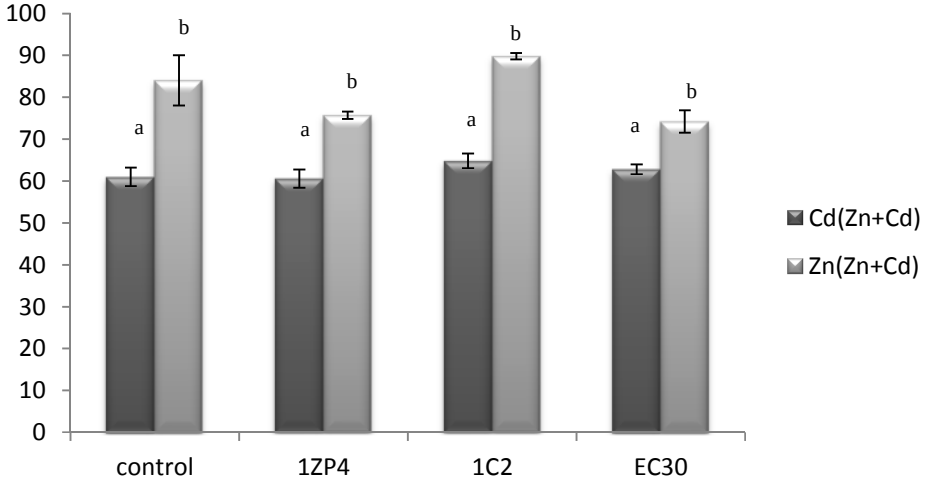
Results are expressed as mean ± S.D. (n = 3). Averages presented considered removal efficiencies observed for the 3 rounds.

Figure 1: Zn and Cd levels in the combined outlet (Zn+Cd) in the alginate matrix with different bacteria applications (mg/L)

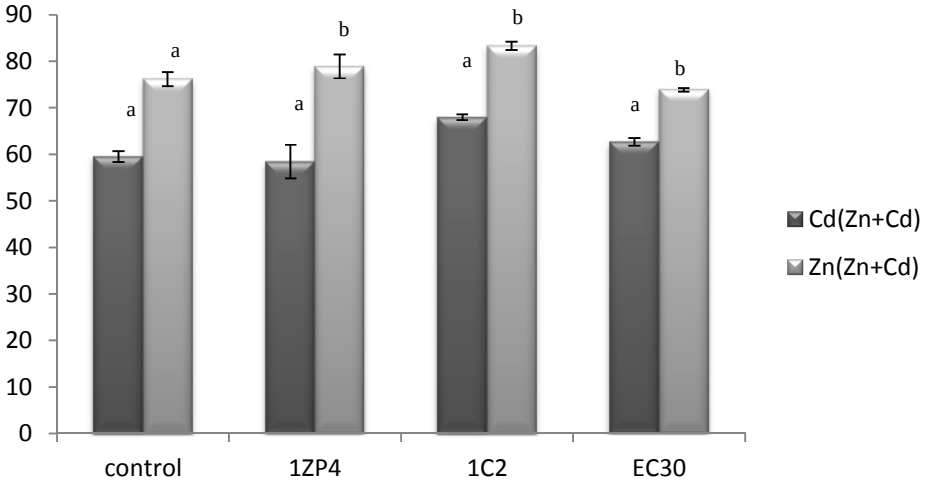
round 1



round 2



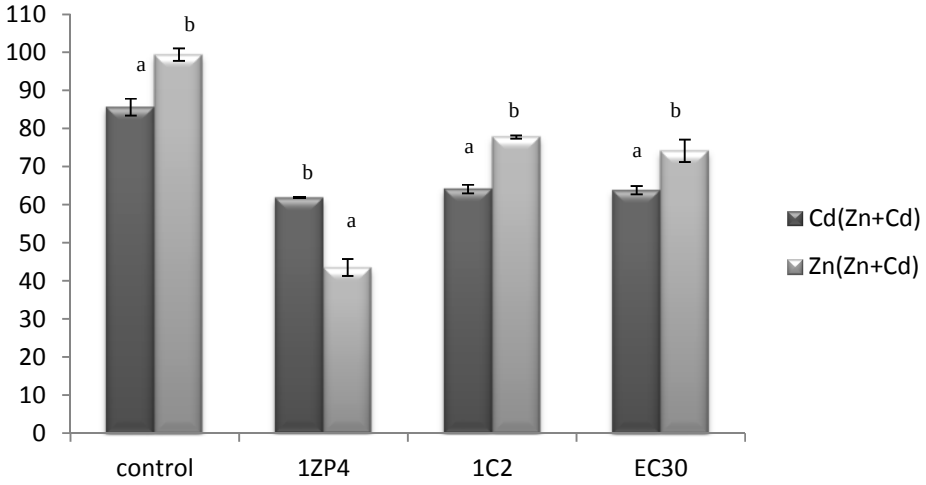
round 3



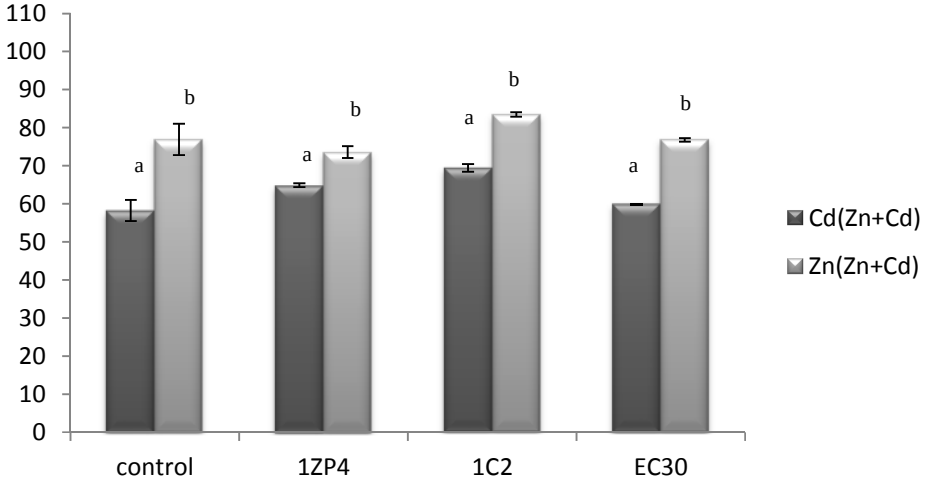
Results are expressed as mean ± S.D. (n = 3). Means for the same bacterial treatment in each round with different letters are significantly different from each other ($P < 0.05$) according to the t-test.

Figure 2: Zn and Cd levels in the combined outlet (Zn+Cd) in the pectate matrix with different bacteria applications(mg/L)

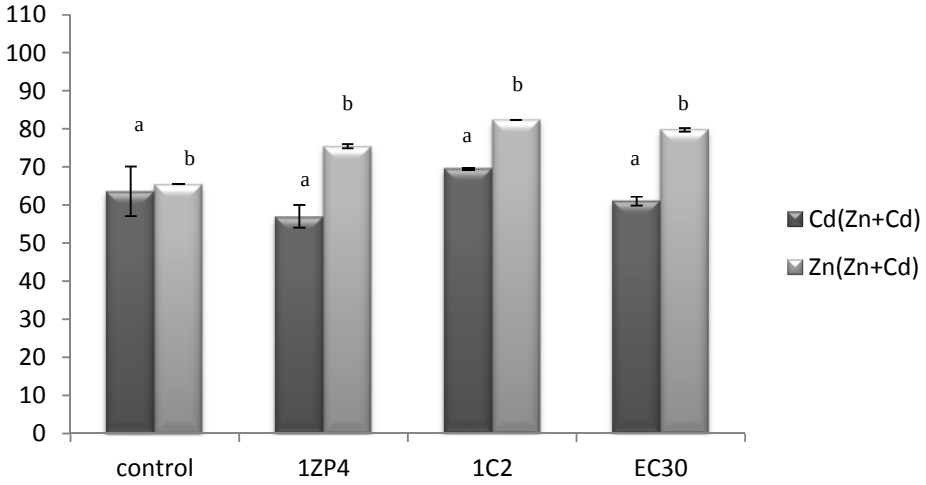
round 1



round 2



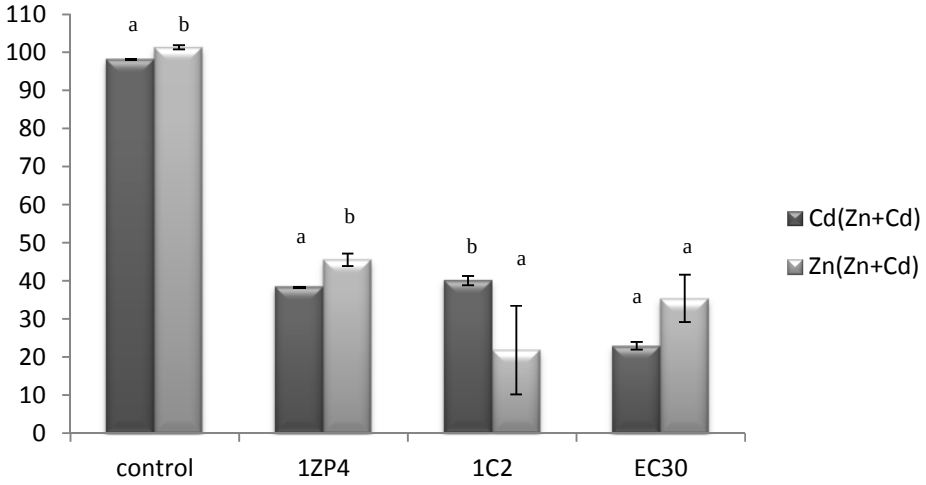
round 3



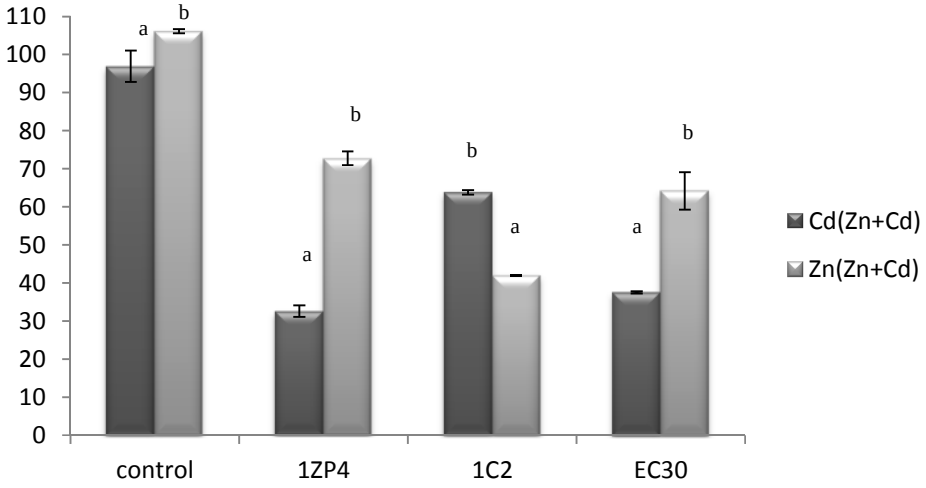
Results are expressed as mean ± S.D. (n = 3). Means for the same bacterial treatment in each round with different letters are significantly different from each other ($P < 0.05$) according to the t-test.

Figure 3: Zn and Cd levels in the combined outlet (Zn+Cd) in the polymer matrix with different bacteria applications (mg/L)

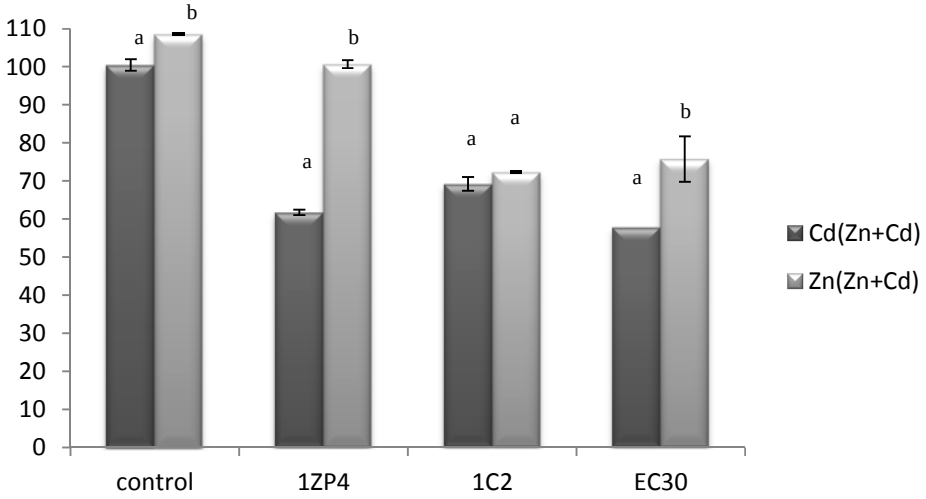
round 1



round 2



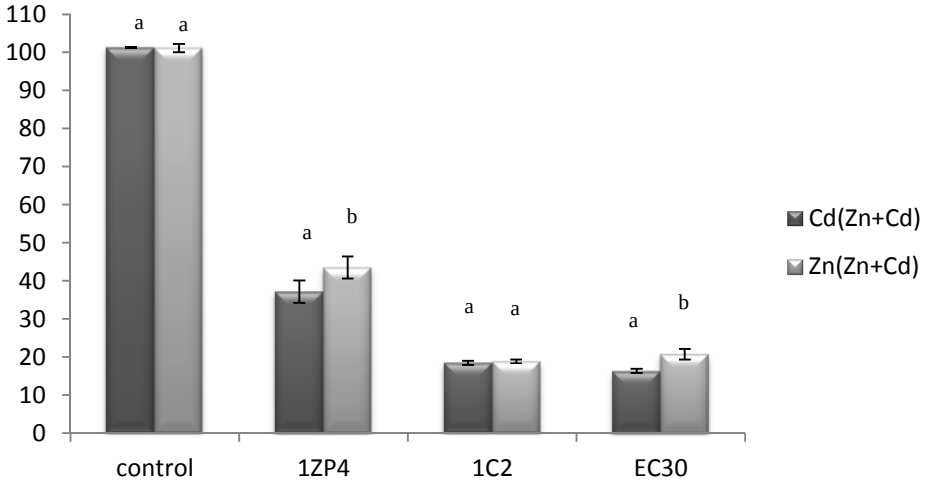
round 3



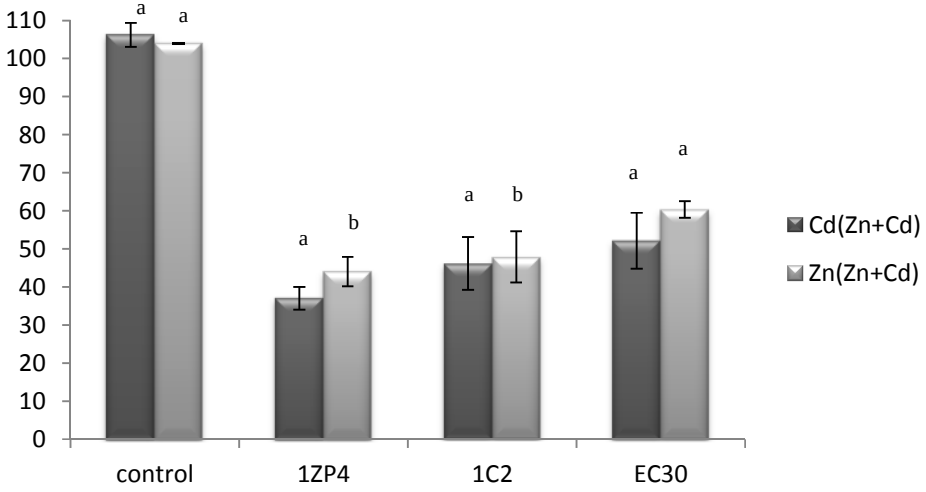
Results are expressed as mean ± S.D. (n = 3). Means for the same bacterial treatment in each round with different letters are significantly different from each other ($P < 0.05$) according to the t-test.

Figure 4: Zn and Cd levels in the combined outlet (Zn+Cd) in the incubated polymer matrix with different bacteria applications (mg/L)

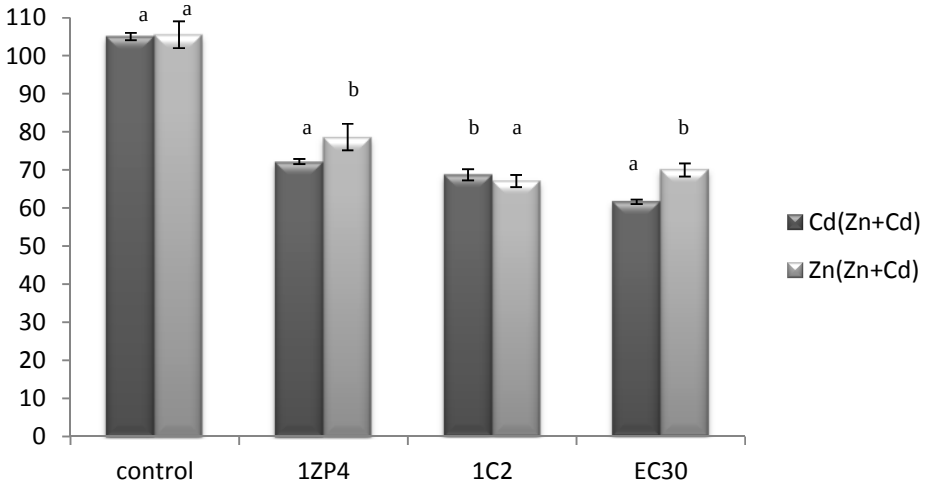
round 1



round 2



round 3



Results are expressed as mean ± S.D. (n = 3). Means for the same bacterial treatment in each round with different letters are significantly different from each other ($P < 0.05$) according to the t-test.

Removal of heavy metals using different polymer matrixes as support for bacterial immobilisation

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Carlos Pires, Ana P.G.C. Marques, António Guerreiro, Naresh Magan, Paula M.L. Castro, Removal of heavy metals using different polymer matrixes as support for bacterial immobilisation, Journal of Hazardous Materials, Volume 191, Issues 1–3, 15 July 2011, Pages 277–286. http://dx.doi.org/10.1016/j.jhazmat.2011.04.079