

Phosphorus Sorption Capacity of Filter Materials Used for On-site Wastewater Treatment Determined in Batch Experiments—A Comparative Study

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Increasing numbers of filter materials have been proposed as suitable media for P removal in on-site wastewater treatment systems. The phosphorus sorption capacity (PSC) of the material can be estimated in batch experiments and is commonly used as the criterion for material selection. However, there is no standard procedure and batch experimental parameters are arbitrarily established, thus leading to difficulties in comparing the results. The main parameters affecting the batch adsorption system are the form and amount of material, material-to-solution ratio, nature, pH and initial concentration of P solution, contact time, agitation, and temperature. This paper critically reviews a number of relevant studies that used batch experiments to estimate the PSC of different filter materials. The nature and form of the materials vary significantly and there is broad variation in the batch experimental parameters set in the selected studies. Analysis of the data from selected studies showed a relationship between particle size or pH of the material and its PSC. The initial P concentration of the solution and the material-to-solution ratio in the batch system were found to be correlated with the estimated PSC, suggesting that batch parameters have a great influence on the results. Based on the analysis of the selected studies, the difficulties of using batch experiments are outlined, recommendations for batch experiment procedure are suggested and a classification system for filter materials according to their PSC and particle size is presented.

PHOSPHORUS in high concentrations is a major source of pollution for natural waters. Agricultural run-off is an important diffuse source of P, while wastewater discharges are the main point sources of P and in fact account for more than 50% of P pollution in Europe (Farmer, 2001; Valsami-Jones, 2004). Urban wastewater treatment has improved significantly in recent decades, but P removal from wastewater in rural areas has not been equally successful (Vymazal et al., 1998). Rural wastewater treatment relies on on-site treatment systems such as infiltration beds, constructed wetlands or filter wells, where soils or filter materials are responsible for P retention (Richardson, 1985; Kadlec and Knight, 1996; Hedström, 2006; Jantrania and Gross, 2006; Shilton et al., 2006). However, commonly used materials such as gravel and sand have a limited P retention ability and, to improve the P removal efficiency, the incorporation of materials with a high affinity for P is necessary. For this reason, research on alternative materials has become a priority for scientists during the past decade. A large number of filter materials with the capacity to remove P from wastewater are described in the literature (e.g., Johansson Westholm, 2006). Some of these materials are considered for use as soil amendments in agriculture (Hyllander et al., 2006; Cucarella et al., 2008).

The mechanism of P retention involves sorption reactions and the efficiency depends on the physical and chemical properties of the material (Mann, 1996; 1997; Zhu et al., 1997; Sakadevan and Bavor, 1998; Drizo et al., 1999). The P removal efficiency can be estimated in batch experiments and the results may be used as criteria for the selection of the most suitable material and to predict the lifetime/longevity of a filter system using such material (Nair et al., 1984; Drizo et al., 2002). The experimental data must be properly interpreted to correctly evaluate the ability of the material to remove P. This ability is generally expressed in terms of PSC. However, batch experimental parameters are arbitrarily set and the results are often noncomparable and misleading (Drizo et al., 2002). Previous work has indicated the difficulty of comparing the PSCs of different materials based on batch experiments (Drizo et al., 2002; Johansson Westholm, 2006). The reason for this is that batch parameters have a large influence on P sorption. A number of equations have been used to describe P sorption (Olsen and Watanabe, 1957; Barrow, 1978; Sposito, 1989; McBride, 1994; Stumm and Morgan, 1996; McGechan and Lewis, 2002),

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Abbreviations: BFS, blast furnace slag; CW, constructed wetland; EAF, electric arc furnace; LECA, light expanded clay aggregates; LWA, light weight aggregates; PSC, phosphorus sorption capacity.

among the most popular being the Langmuir and Freundlich equations. The Langmuir equation has been widely used and it has the advantage that it gives an estimate of the maximum PSC. On the other hand, the application of this equation has many limitations and the results must be viewed with caution (Barrow, 1978; Arias et al., 2001; Drizo et al., 2002).

This paper compares and closely analyzes several studies using batch experiments to estimate the PSC of filter materials (Mann and Bavor, 1993; Cheung et al., 1994; Zhu et al., 1997; Sakadevan and Bavor, 1998; Drizo et al., 1999; Johansson, 1999; Zhou and Li, 2001; Drizo et al., 2002; Kostura et al., 2005; Li et al., 2006; Xu et al., 2006; Ádám et al., 2007a; Boujelben et al., 2008). The studies were selected for their relevance to the subject and repercussions on later research. The selected studies cover a large variation in filter materials. The objective was to critically review the selected studies and assess the validity of the results to evaluate: (i) the reliability of estimating the PSC of filter materials using batch experiments, (ii) the relationship between the properties of materials and their PSC, (iii) the influence of batch parameters on P sorption, and (iv) the limitations of using batch experiments. Based on this evaluation, the aim of the paper is to produce a set of recommendations for batch experiment procedures and to suggest a classification system for filter materials according to their PSC and particle size.

Filter Materials

Filter materials used for P removal from wastewater are characterized by a high affinity for P and appropriate hydrological properties. These materials are also called substrates and can be classified into three groups: natural materials, industrial by-products, and manufactured products (Johansson Westholm, 2006). Non-reactive or inert materials such as sand and gravel have long been suggested as suitable filter media for P removal from wastewater (Mann and Bavor, 1993; Zhu et al., 1997; Arias et al., 2001). Reactive materials are referred to as adsorbents or sorbents and specifically interact with targeted chemical species (e.g., phosphate ions). In surface chemistry, adsorption can be defined as the net accumulation of matter at the solid-water interface (Stumm and Morgan, 1996). The term sorption was defined by McBride (1994) as a continuous process that ranges from adsorption to precipitation reactions. This term is very convenient when the chemical processes governing the interaction are not fully known, as is the case for the reaction of phosphate ions with the different reactive materials studied and used for P removal from wastewater.

The ability of a material to remove P from wastewater depends on its physical and chemical properties. The shape, particle size, and porosity of the grains or aggregates define their specific surface area, and, generally, the smaller the particle size the larger the surface area to undergo P sorption (Nair et al., 1984; Mann, 1996; Zhu et al., 1997). The chemical composition together with the pH determines the affinity or reactivity and the strength of the interaction. The P removal efficiency of a material is closely related to the content of Al, Ca, and Fe, as well as the pH (Grubb et al., 2000; Johansson and Gustafsson, 2000; Arias et al., 2001, 2006; Khadhraoui et al., 2002), and

the abundance of these elements times the surface area defines the number of sorption sites. Thus, the sorption capacity depends on both the number and affinity of sorption sites.

The materials studied in the selected papers are bauxite, bentonite, black oxide, calcareous soils, limestone, FILTRALITE/Filtralite P (derived from LECA) (maxit as, Alnabru, Oslo, Norway), Fe-coated sand and brick, fly ash, gravel, LECA (Light Expanded Clay Aggregates), Lehigh Cement VA, opoka, red mud (bauxite residue), sand, shale, shell sand, slags including blast furnace slag (BFS), electric arc furnace (EAF) steel slag and steel furnace slag (SFS), spodosol, UTELITE (Utelite Corp., Coalville, UT) and zeolites. Table 1 shows the particle size, pH, and chemical composition of these materials (the chemical composition is presented as percentages from data in the original papers).

Calcium-rich natural materials are of interest due to the strong interaction of P with some Ca compounds. Among these, opoka, a sedimentary deposit with a high content of calcium carbonate and silica, has shown promising results when heated to more than 900°C (Brogowski and Renman, 2004). Although many of the materials investigated are of natural origin, the greatest interest has been devoted to industrial by-products such as slag materials and fly ash (Yamada et al., 1986; Mann and Bavor, 1993; Sakadevan and Bavor, 1998; Drizo et al., 1999, 2002; Johansson, 1999; Xu et al., 2006). Slags are by-products of steel making works and can vary widely in form and composition depending on the manufacture process. They usually contain elevated concentrations of CaO and relatively high concentrations of Al and Mg oxides (Table 1). Electric arc furnace (EAF) steel slag is rich in both Ca and Fe (Drizo et al., 2002). Fly ash is the by-product of the coal combustion process and the composition of fly ash can vary strongly depending on the process conditions. Among the samples of fly ash investigated, there is a wide range of pH values and chemical composition (Table 1). The group of lightweight aggregates (LWA) has also been widely studied by many authors (Zhu et al., 1997; Drizo et al., 1999; Johansson, 1999; Ádám et al., 2007a) and the most popular in this group are LECA and Filtralite. The latter is the product of processing LECA to improve its PSC.

The increasing demand for efficient materials for P removal from wastewater has led to the appearance of new derivatives and the manufacture of novel products. Some of the most relevant are the commercial products Polonite (derived from opoka) (Biopotech AB, Hallstavik, Sweden) and Filtra P (Nordkalk, Pargas, Finland) (Renman et al., 2004; Cucarella et al., 2007; Gustafsson et al., 2008). Both of these have relatively high pH values and a high content of CaO. In addition to P removal, these products have shown promising results as P fertilizers on acid soils (Cucarella et al., 2008). These materials are not included in this paper due to the lack of studies using them in batch experiments.

Batch Adsorption Systems

Contact adsorption systems have long been used to remove pollutants from wastewater (McKay, 1996). Such systems consist of a batch reactor in which the effluent (e.g., wastewater) and the adsorbent are intimately mixed for a certain time to enable the system to approach equilibrium. The time required for the adsorbent

Table 1. Particle size, pH, and chemical composition or origin of the filter materials.

Material	Diameter	pH	Composition or origin (dominant elements)	Reference
	mm			
Blast furnace slag	9.5–19	n.d.†	42%CaO, 6%MgO, 0.3%Fe ₂ O ₃ , 15%Al ₂ O ₃	Mann and Bavor, 1993
Fly ash	<0.5	n.d.	0.2%CaO, 0.3%MgO, 3.5%Fe ₂ O ₃ , 70.7%Al ₂ O ₃	
Granulated slag	0.08–4.75	n.d.	n.d.	
Gravel (R/G)	5–10/3–5	n.d.	0.08/0.01%MgO, 0.8/0.4%Fe ₂ O ₃ , 0.4/4.5%Al ₂ O ₃	
Black oxide	<2	5.2	Taken from mineral sands processing industry	Cheung et al., 1994
Fly ash (I/II)	<2	5.9/7.6	From Kwinana (I) and Pillar Point (II) power station	
Red mud gypsum	<2	8.5	Red mud + 5% gypsum	
FILTRALITE	<2	10.4/10.3	24.8/31.0%Ca, 16.2/12.5%Mg, 3.0/3.4%Fe, 8.5/11.9%Al	Zhu et al., 1997
LECA (I/II)	<2	9.4/10.5	5.6/8.5%Ca, 3.3/3.6%Mg, 2.2/3.9%Fe, 6.0/6.5%Al	
LehighCementVA	<2	9.2	34.1%Ca, 7.3%Mg, 3.0%Fe, 11.6%Al	
Sand (I/II)	<2	5.8/5.1	4.2/3.9%Ca, 4.7/4.7%Mg, 6.1/3.0%Fe, 10.1/7.4%Al	
UTELITE	<2	10.1	45.6%Ca, 9.3%Mg, 3.3%Fe, 14.7%Al	
Blast furnace slag	n.d.	n.d.	38–43%CaO, 5–8%MgO, < 1.3%FeO, 13–16% Al ₂ O ₃ , 32–37%SiO ₂	Sakadevan and Bavor, 1998
Steel furnace slag	n.d.	n.d.	35–45%CaO, 7–12%MgO, 20–30%FeO, 1–5%Al ₂ O ₃ , 10–15%SiO ₂	
Zeolite (70% clinoptilolite)	n.d.	n.d.	2.1%CaO, < 1%MgO, < 1%FeO, 13%Al ₂ O ₃ , 66%SiO ₂	
Bauxite	6–13	5.9	Al and Fe oxides	Drizo et al., 1999
Fly ash	0–4	8.3	SiO ₂ , Al ₂ O ₃ and Fe oxides	
LECA	6–14	8.2	Light expanded clay aggregates	
Limestone	6–14	7.8	Calcium carbonate	
Shale	2–13	4.5	Limestone derived	
Zeolite	6–13	6.5	Aluminium-silicate	
Amorphous slag (F/C)‡	0–0.125/ 0.25–4	10.6/10.3	35%CaO, 13.4%MgO, 10.6%Al ₂ O ₃ , 36.2%SiO ₂	Johansson, 1999
Crystalline slag (F/C)	0–0.125/ 0.25–4	10.2/10.3	35%CaO, 13.4%MgO, 10.6%Al ₂ O ₃ , 36.2%SiO ₂	
Fly ash	n.d.	4.9	23.7%CaO, 23.0%Fe ₂ O ₃ , 12.5%Al ₂ O ₃ , 22.1%SiO ₂	
Limestone	0.25–2	8.9	53%CaCO ₃ , 4%MgO	
Opoka	0.125–2	8.3	50%CaCO ₃ , 4%MgO, 5%Fe ₂ O ₃ , 10.6%Al ₂ O ₃ , 36.2%SiO ₂	
Spodosol	n.d.	5.4	3.4%Fe ₂ O ₃ , 13%Al ₂ O ₃	
Calcareous soils	n.d.	7.3–8.2	High CaCO ₃ content	Zhou and Li, 2001
Limestone	n.d.	n.d.	n.d.	
Electric arc furnace steel slag	2.5–10	n.d.	21.7%Ca, 7.9%MgO, 24.3%Fe ₂ O ₃ , 2.5%Al ₂ O ₃ , 6.4%Si	Drizo et al., 2002
Amorphous slag	0–0.1	n.d.	32.1–33.6%CaO, 15.1–15.6%MgO, 0.2–0.7%Fe ₂ O ₃ , 6.2–7.2%Al ₂ O ₃ , 40.2–40.4%SiO ₂	Kostura et al., 2005
Crystalline slag	0–0.1	n.d.	38.0%CaO, 13.7%MgO, 0.2%FeO, 6.6%Al ₂ O ₃ , 38.6%SiO ₂	
Fly ash	<0.15	9.37	2.7%CaO, 1.5%MgO, 7.3%Fe ₂ O ₃ , 25.4%Al ₂ O ₃ , 56.4%SiO ₂	Li et al., 2006
Red mud	<0.15	11.70	46%CaO, 1.2%MgO, 12.8%Fe ₂ O ₃ , 6.9%Al ₂ O ₃ , 19.1%SiO ₂	
Bentonite	<0.25	3.03	1.8%CaO, 3.1%MgO, 0.5%Fe ₂ O ₃ , 16.7%Al ₂ O ₃ , 67.5%SiO ₂	Xu et al., 2006
Fly ash	<0.25	12.1	3.1%CaO, 0.9%MgO, 6.9%Fe-oxides, 27.8%Al ₂ O ₃ , 50.4%SiO ₂	
Furnace slag	0–5	12.3	Furnace bottom ash with similar composition to fly ash	
Sand (I-IV)	0–5	6.7–8.1	32.5–56.3%Ca, 0.9–1.6%Mg, 2.3–6.3%Fe, 1.3–2.0%Al	
Filtralite P	0.5–4	10.7	3.1%Ca, 0.7%Mg, 0.6%Fe, 2.0%Al	Ádám et al., 2007
Shell sand	3–7	8.82	32.8%Ca, 1.4%Mg, 0.05%Fe, 0.03%Al	
Fe-coated sand	0.7–1	n.d.	0.3%Fe	Boujelben et al., 2008
Fe-coated brick	0.8–2	n.d.	0.5%Fe	

† n.d., no data.

‡ F, fine; C, course.

and liquid to come to substantial equilibrium depends primarily on the concentration and particle size of the solid, the viscosity of the liquid, solute concentration and the extent of agitation.

For a single-stage adsorption system, the mass balance at equilibrium is:

$$V \times (C_0 - C_1) = M \times S \quad [1]$$

where V (L⁻¹) is the volume of solvent having a solute concentration of C (g L⁻¹) and M (kg) is the amount of adsorbent. The solute concentration in solution is reduced

from C_0 to C_1 (g L⁻¹) and becomes adsorbed to the solid at a concentration S (g kg⁻¹).

Similar batch adsorption systems have been used at laboratory scale to assess the ability of soils or materials to attract phosphate ions. This experimental set-up is commonly called a batch experiment and consists of placing a fixed amount of the material with a mass M (g) in a beaker or Erlenmeyer flask containing a volume V (L⁻¹) of a prepared P solution at one of a range of increasing concentrations. The samples are shaken in a rotator at speed v (rpm) for a time t (h) at temperature T (°C). The difference between the initial and final P concentrations in

solution at equilibrium (assumed to be reached at time t), C_0 and C_{eq} , respectively, is assumed to be sorbed to the material. The concentration of P is determined by colorimetric methods (e.g., Murphy and Riley, 1962). The amount of P sorbed to the material (S) is expressed in unit mass P (g) per unit mass of the material (kg) and is calculated as:

$$S = [(C_0 - C_{eq}) \times V] / M \quad [2]$$

Table 2 shows the batch experiment parameters used in the selected studies. There are large discrepancies for most of the parameters used. Batch experiment data are primarily used to compare the P removal efficiency of different materials and thus have the aim of determining which material will work best under certain conditions. The efficiency can be expressed as the percentage of P removal or as the PSC, which equals the maximum observed or correlated value of S in the experiment. The PSC of the materials investigated in the selected studies is shown in Table 3. There is a four orders of magnitude difference in P sorption for the materials investigated; the lowest values are for gravel and sand, while the highest are reported for red mud, fly ash, and blast furnace slag.

The relationship between the PSC of the filter materials and their particle size and pH is shown in Fig. 1 and 2, respectively. The graphical representation shows that smaller particle diameters usually have a higher P sorption and coarser fractions have lower. The relationship between P sorption and pH shows a tendency to higher sorption with increasing pH values. While low pH values indicate low P sorption, higher values do not necessarily imply high P sorption. The data were insufficient to plot the relationship with the chemical composition. However, many studies have shown a correlation between P sorption and the content of Al and Fe (Baker et al., 1998; Sakadevan and Bavor, 1998; Grubb et al., 2000; Arias et al., 2006) and Ca (Zhu et al., 1997; Arias et al., 2001; Khadhraoui et al., 2002). Another study showed a good correlation between the Langmuir maxima and the content of Ca and Mg in the material (Del Bubba et al., 2003). Seo et al. (2005) showed that addition of Ca to a filter material improved its PSC more than addition of Al, Fe, and Mg.

However, the PSC is relative because it is estimated under different conditions. The data strongly depend on the experimental procedure itself and the most important parameters are the form and amount of the material, the material-to-solution ratio, the nature, pH and initial concentration of the P solution, the contact time, agitation, and temperature (Barrow, 1978; Nair et al., 1984).

Form and Amount of Material

In a batch experiment, the material should be used in the same form as is expected to be used in field conditions (Drizo et al., 1999). This can range from the size of small particles in soils (common in infiltration beds and wetland systems with slow flow) to larger sand grains, gravel stones or even coarser materials (used in rapid filter treatment systems such as filter wells). In the selected studies, materials with different particle diameters ranging from microns to millimeters (Table 1) have been tested in batch

experiments under similar conditions (Table 2). However, the P removal efficiency of a material strongly depends on its particle size, and therefore comparison of materials with significantly different particle size may yield irrelevant results. For example, the P removal efficiency of fly ash and slag has been shown to decrease with an increase in particle size of only 0.05 to 0.3 mm (Agyei et al., 2002). The solution pH at equilibrium has also been found to be dependent on the particle size of slags (Kostura et al., 2005). On the other hand, Dunne et al. (2008), recently reported that there were no relationships between particle size and P sorption for several materials investigated, except for sand.

A fixed amount of the material is used in a batch experiment and this amount should represent all of its composition. Authors do not agree on the optimum amount of material, although most of them have used about 3 g (Table 2). Drizo et al. (1999, 2002) used much larger quantities (20 and 35 g) and in the first study obtained relatively similar sorption capacity values (0.46–0.86 g P kg⁻¹, Table 3) for six materials with clear differences in composition, pH, and particle size (Drizo et al., 1999, Table 1). The sorption isotherms did not show any saturation within the range of P concentrations applied. Mann and Bavor (1993) also used a large amount (20 g) to study the P removal by industrial by-products and obtained rather low P sorption capacities for fly ash and slags compared with the results obtained in later work (Cheung et al., 1994; Sakadevan and Bavor, 1998). On the other hand, Li et al. (2006) used a very small amount (0.1 g), which may not be sufficiently representative. The amount of the material used in a batch experiment is closely related to the volume of the solution used so that an adequate material-to-solution ratio is defined.

Material-to-Solution Ratio

Based on a substrate-solute equilibrium, it seems clear that a variation in the material-to-solution ratio may alter either the equilibrium constant or the complex concentration by reaching a new equilibrium. Logically, smaller ratios may lead to higher concentrations of P sorbed to the material, that is, a higher percentage of P removal, but not necessarily a higher sorption capacity (P sorbed per unit mass of material). This has been observed with shell sand (Søvik and Kløve, 2005) and Filtralite P (Ádám et al., 2007b). Those studies concluded that the smaller the ratio, the longer the time needed to reach equilibrium. It is also important to mention that the smaller the material-to-solution ratio, the greater the difference between batch tests and field conditions, where a much larger amount of the material is in contact with the solution (depending on bulk density, porosity, hydraulic load, etc.). In this respect, the batch test can underestimate P retention ability if small material-to-solution ratios are used. On the other hand, the whole surface of the material is exposed in batch solution, while it is not in full-scale applications (constructed wetland, filter well) where the material is packed, thus reducing the total surface area for P interactions. In such cases, batch experiment data can overestimate the P removal ability of the material and the smaller the particle size, the larger the error.

The material-to-solution ratio used in the batch experiments from the selected papers varies notably (Table 2). Ra-

Table 2. Batch experiment parameters used in the selected studies.

Materials	Mass	Volume	Material-to-solution ratio	Electrolyte	P concentration	Time	Rotation	Temperature	Estimation of		Reference
									mg L ⁻¹	h	
Blast furnace slag, Fly ash, Gravel	20	40	1:2	KCl (0.01 mol L ⁻¹)	5–100	24 (30)	1500	25	Langmuir, Freundlich	Mann and Bavor, 1993	
Black oxide, Fly ash, Red mud gypsum	5	25	1:5	KCl (0.01 mol L ⁻¹)	0–800	25	n.d.†	25	Langmuir, Freundlich	Cheung et al., 1994	
FILTRALITE, LECA, Lehigh Cement VA, Sand, UTELITE	8	200	1:25	Water	0–320	24	n.d.	22	P-sorbed at highest initial P concentration	Zhu et al., 1997	
Blast furnace slag, Steel furnace slag, Soil, Zeolite	3	30	1:10	KCl (0.01 mol L ⁻¹)	10–1000 (10,000)	48	100	25	Langmuir	Sakadevan and Bavor, 1998	
Bauxite, Fly ash, LECA, Limestone, Shale, Zeolite	20	?	1:25	CaCl ₂ (0.01 mol L ⁻¹)	2.5–40	24	60	21	Langmuir; Column test (35–45 mg P L ⁻¹) at 3 L d ⁻¹	Drizo et al., 1999	
Fly ash, Furnace slag, Limestone, Opoka, Spodosol	1	50	1:50	NaNO ₃	5–25	24	70	Room (?)	P-sorbed at highest initial P concentration	Johansson, 1999	
Calcareous soils, Limestone	3	30	1:10	KCl (0.05 mol L ⁻¹)	0.1–1250	24	n.d.	Room (?)	Langmuir, Freundlich; One-point isotherm	Zhou and Li, 2001	
Electric arc furnace steel slag	35	700	1:20	Water	1–320	24	175	24	Langmuir; Column test (350–400 mg P L ⁻¹) at 1.73 L d ⁻¹	Drizo et al., 2002	
Amorphous slag, Crystalline slag	0.5	100	1:200	Water	50–500	150 (1min)	0 (200)	20	Langmuir, Freundlich	Kostura et al., 2005	
Fly ash, Red mud	0.1	20	1:200	KCl (0.01 mol L ⁻¹)	0.31–3100	4	180	25	Langmuir, Freundlich	Li et al., 2006	
Bentonite, Fly ash, Furnace slag, Sand, Soil	2	40	1:20	NaCl (0.01 mol L ⁻¹)	10–100 10–1000	24	200	25	Langmuir	Xu et al., 2006	
Filtralite P, Shell sand	3	90	1:30	Water	0–480	24	n.d.	n.d.	P-sorbed at highest P concentration; Column test (5–10 mg P L ⁻¹) at 4.5–5.5 L d ⁻¹	Adám et al., 2007	
Fe-coated sand, Fe-coated brick	5	250	1:50	NaOH HNO ₃	5–30	2	n.d.	20	Langmuir, Freundlich	Boujelben et al., 2008	

† n.d., no data.

Table 3. Phosphorus sorption capacity of the materials investigated.

Material	P sorption capacity	Conclusions/Practical application	Reference
	g P kg ⁻¹		
Blast furnace slag	0.42	The industrial waste substrate (Blast furnace slag, Fly ash) may have potential as substrate suited to phosphorus removal in a constructed wetland (CW) system.	Mann and Bavor, 1993
Fly ash	0.26		
Granulated slag	0.16		
Gravel (R/G)	0.03/0.05		
Black oxide	0.89	The results suggest that alkaline fly ash may be a promising amendment for coarse sand bed to enhance P removal. Batch experiments should only be considered as an initial estimate of the PSC.	Cheung et al., 1994
Fly ash (I/II)	1.19/3.08		
Red mud gypsum	5.07		
FILTRALITE	1.39/2.21	The PSC of different LWA products and sands varies by two orders of magnitude. The results are not sufficient for predicting the substrate life-length as a P-sink for a CW system.	Zhu et al., 1997
LECA (I/II)	0.16/0.57		
Lehigh Cement VA	2.91		
Sand (I/II)	0.43/0.44		
UTELITE	3.46		
Blast furnace slag	44.25	CW systems with 60 cm substrate depth, surface area of 6.25 m ² and hydraulic load rate of 1250 L d ⁻¹ with 8 mg P L ⁻¹ would last for 11 yr for zeolite and 58 yr for blast furnace slag.	Sakadevan and Bavor, 1998
Steel furnace slag	1.43		
Zeolite	2.15		
Bauxite	0.61	The results are not sufficient to forecast accurately the lifetime of a CW system. Batch experiments can be used to obtain an initial estimate. Based on column results, a lifetime of 7 yr is suggested for shale in a 5 m ² × 0.6 m CW with 440 g P yr ⁻¹ load.	Drizo et al., 1999
Fly ash	0.86		
LECA	0.42		
Limestone	0.68		
Shale	0.65		
Zeolite	0.46		
Amorphous slag (F/C)†	0.42/0.15		
Crystalline slag (F/C)	1.00/0.65		
Limestone	0.25		
Opoka	0.10		
Spodosol	1.00		
Calcareous soils	0.59–5.55	P sorption values estimated from the one-point isotherm are very similar to those calculated from the Langmuir isotherm.	Zhou and Li, 2001
Limestone	0.01 (0.42)		
Electric arc furnace steel slag	3.93 (batch) 1.35–2.35 (column)	P adsorption capacity derived from batch experiments can vary by several orders of magnitude. Life expectancy of a full-scale system (Drizo et al., 1999) would be 13–37 yr (based on column results).	Drizo et al., 2002
Amorphous slag (I/II)	6.47/8.50	P sorption by slags can be described by the Langmuir equation. Acidification, neutralizing and sorption capacities are well correlated.	Kostura et al., 2005
Crystalline slag	18.94		
Fly ash	63.22 (78.44)	Both substrates are inexpensive and can be economically used for the removal of P from wastewater. The PSC of a sorbent can be improved by 'activation' either thermally or with acids (results in brackets).	Li et al., 2006
Red mud	113.87 (345.50)		
Bentonite	0.93	Lifetime of CW can be estimated using the Langmuir P sorption maximum. Furnace slag shows promise as a substrate for CW and may have a potential lifetime of 22 yr. The Langmuir isotherm may provide a quick screening tool but full-scale supplementary research is needed to accurately forecast PSC.	Xu et al., 2006
Fly ash	8.81		
Furnace slag	8.89		
Sand (I-IV)	0.13–0.29		
Filtralite P	2.50	Batch and column studies suggest that shell sand has a more durable PSC than Filtralite P due to a persistently higher concentration of Ca.	Ádám et al., 2007
Shell sand	9.60		
Fe-coated sand	1.50	Fe-coated sorbents can efficiently remove P from contaminated water. Langmuir and Freundlich equations fitted experimental data satisfactorily.	Boujelben et al., 2008
Fe-coated brick	1.75		

† F, fine; C, coarse.

tios of 1:10 or 1:20 are quite common because most batch experiments are based on the methods used in soil science, where a soil-to-solution ratio of 1:20 is recommended. Some studies use larger ratios (Mann and Bavor, 1993; Cheung et al., 1994) while others use batch systems with smaller ratios of up to 1:200 (Kostura et al., 2005; Li et al., 2006), which may be too small. The relationship between the estimated PSC and the

material-to-solution ratio is shown in Fig. 3. It can be observed a clear tendency for higher sorption with decreasing ratios.

Phosphorus Solution

The nature of the P solution used in batch experiments differs considerably among studies. Potassium dihydrogen phosphate is generally used and the most common solvent is water. How-

ever, many authors use a certain concentration of electrolyte to maintain a constant ionic strength. The type and concentration of the supporting electrolyte affects the interactions in solution. The presence of ions other than phosphate ions and the material under investigation may interfere with the sorption process and, consequently, the sorption capacity. One study has shown that water may be the most appropriate solvent for such experiments (Janyes et al., 1999). Drizo et al. (2002) substituted the CaCl_2 solvent used previously in a similar experiment (Drizo et al., 1999) with water (Table 2). Calcium ions in solution can quite probably precipitate with phosphate ions and therefore calcium compounds should not be used. Many studies use KCl or NaCl as background electrolyte (Table 2). It is important to emphasize that using artificial P solutions creates an ideal system in which P is isolated, as opposed to what happens in wastewater, where many other species compete for sorption sites on the material and P can attach to other compounds or undergo transformations. The PSC of BFS with wastewater has been found to be considerably lower than that estimated with artificial solutions (Hedström and Rastas, 2006). The effect of organic matter on P sorption has been studied by Seo et al. (2005) and Xu et al. (2006) and the presence of organic ligands has been found to inhibit Ca-P precipitation (Van der Houwen and Valsami-Jones, 2001).

The initial P concentration in a batch test can be set in the range in which the material is expected to remove P for a given material-to-solution ratio. Typical concentrations of P in wastewater range from 5 to 10 mg P L^{-1} but they can be as high as 20 or 30 mg P L^{-1} in some cases. It is common to use higher concentrations than those in batch experiments where P sorption isotherms are plotted. However, using much higher concentrations may lead to erroneous results (Drizo et al., 2002). Higher P concentrations give a lower pH value, which certainly influences the sorption reaction. The effect of pH on P sorption has been studied for different materials (Agyei et al., 2002; Li et al., 2006; Boujelben et al., 2008). In addition, very high concentrations create a new and probably different equilibrium situation that might not reflect what happens at lower concentrations. It is known that some materials have a different behavior with low and high P concentrations (Søvik and Kløve, 2005; Ádám et al., 2007a). The initial P concentration has been found to affect the P removal efficiency of fly ash and slag, which increases with higher initial P concentrations (Agyei et al., 2002). Drizo et al. (2002) demonstrated that large differences in applied initial P concentrations significantly affected the estimated P sorption for EAF slag. In another study, the materials shell sand and Filtralite P had opposite efficiencies at low and high initial P concentrations (Ádám et al., 2007a). It is also known that precipitation reactions usually occur at high P concentrations, while adsorption dominates at low concentrations. This has been observed for shell sand by Søvik and Kløve (2005).

There are notable differences in the initial P concentration range in batch solutions used in the selected studies (Table 2). Only in some cases are the values within the range of wastewater P concentrations (Drizo et al., 1999; Johansson, 1999; Boujelben et al., 2008). In most cases, however, concentrations are high or even very high compared to typical P concentrations in wastewater, reaching values more than 1000 mg P L^{-1} (Sakadevan and Bavor,

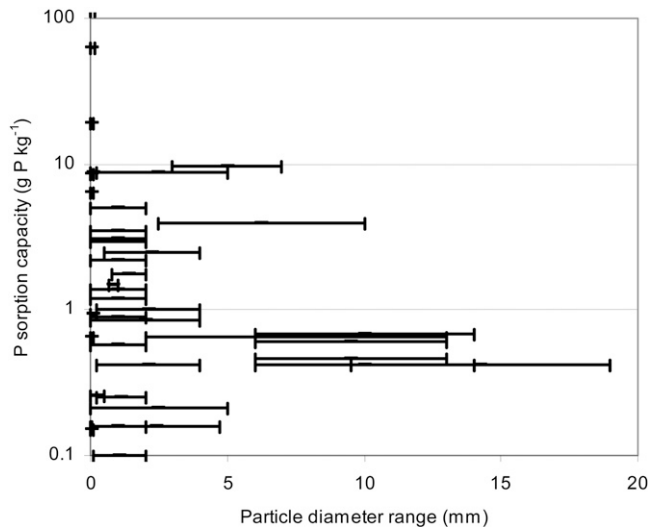


Fig. 1. Relationship between the P sorption capacity (in logarithmic scale) and particle size distribution (bars representing the range) of the materials investigated in the selected studies.

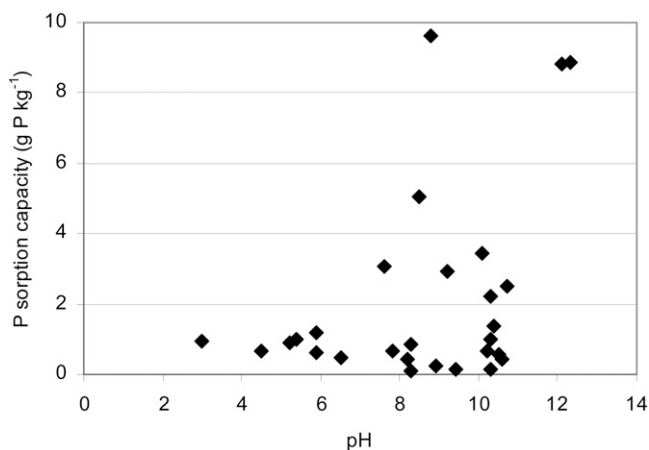


Fig. 2. Relationship between the P sorption capacity and pH of the materials investigated in the selected studies (highest P sorption values excluded).

1998; Zhou and Li, 2001; Li et al., 2006). Special mention has to be made of the elevated concentration of 10,000 mg P L^{-1} used to saturate 3 g of BFS by Sakadevan and Bavor (1998), thus obtaining two orders of magnitude higher P sorption compared with the results from previous work (Mann and Bavor, 1993). The relationship between the estimated PSC and the highest initial amount of P in solution is shown in Fig. 4. There is a clear tendency for higher sorption with increasing P concentration and this tendency seems to be exponential, so that the influence becomes stronger with increasing concentration. The highest values correspond to red mud and fly ash (Li et al., 2006), which were estimated using very high P concentrations of up to 3100 mg P L^{-1} .

Contact Time, Agitation, and Temperature

The contact time should be sufficient to enable the system to reach equilibrium. However, P sorption processes are complicated and usually consist of fast sorption reactions followed by slow processes that are then subdivided into relatively faster and very slow

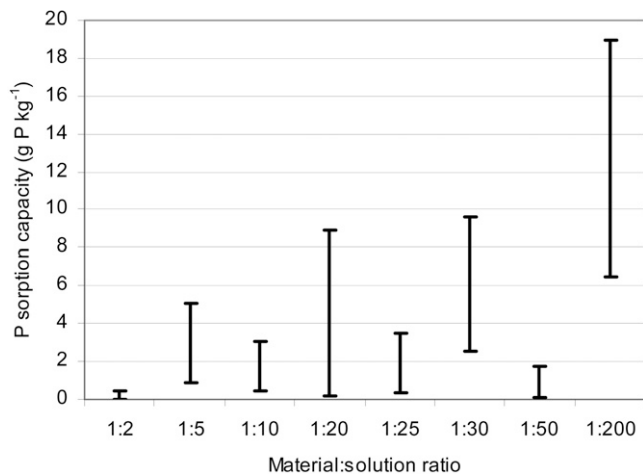


Fig. 3. Phosphorus sorption capacity range (bars indicating the range of different values of sorption capacity) at different material-to-solution ratios used in batch experiments in the selected studies (highest P sorption values excluded).

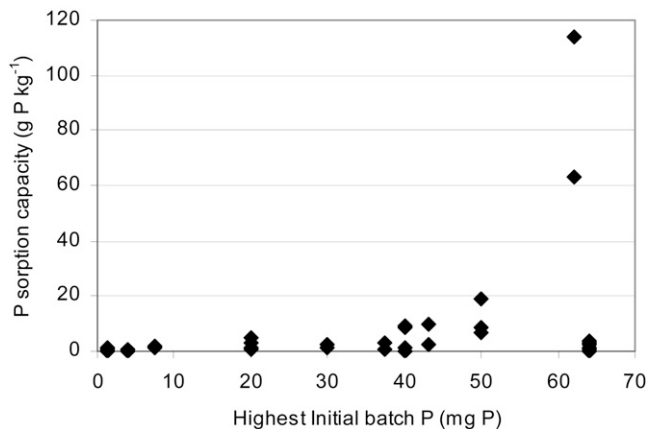


Fig. 4. Relationship between the P sorption capacity of the materials investigated and the amount of initial P at the highest concentration in batch solution in the selected papers.

reactions (Barrow and Shaw, 1975; Van der Zee and Van Riemsdijk, 1991; McGechan and Lewis, 2002). For this reason, equilibrium may be reached after minutes, hours, days, or even months depending on the type of material used, the material-to-solution ratio, initial P concentration, agitation, and temperature.

The time needed to reach equilibrium depends strongly on the particle size of the material. Sorption kinetics studies have shown that fly ash and furnace slag with a very small particle diameter (about 0.03 mm in average) need only 3 to 5 h to reach equilibrium (Agyei et al., 2002). The same time was needed to reach equilibrium for fly ash and red mud with particle diameter <0.15 mm (Li et al., 2006). On the other hand, a study of the P removal efficiency of apatite-containing materials and Graymont limestone with larger particle diameter (2.5–10 mm) showed a two- to threefold increase in P sorption from 24 to 96 h (Bellier et al., 2006). The lowest values were attributed to the lower porosity of the materials in that study. This means that the higher the porosity of a material, the higher the specific surface area and sorption capacity, but at the same time, the longer the contact

time needed to reach equilibrium due to slow P movement processes such as intraparticle diffusion. This is well explained by McKay (1996), who generalizes the sorption process into three steps: (i) external (film) mass transfer of solute molecules (e.g., phosphate ions) from the bulk solution to the sorbent particle surface; followed by (ii) diffusion within the particle internal structure; where (iii) rapid uptake at sorption sites occurs. The first two steps remain the rate-determining steps and, while film transfer takes the first minutes of the process, intraparticle diffusion occurs over several hours. According to this, using the same contact time for small particles and coarser materials may yield noncomparable results. This is the case in the selected papers where materials having different particle diameters (Table 1), in some cases with even two orders of magnitude difference (BFS), are equally mixed with solution during a contact time of 24 h (Table 2). In two cases (Li et al., 2006; Boujelben et al., 2008), contact times are much shorter (4 and 2 h), thus questioning whether equilibrium was reached.

The contact time needed to reach equilibrium in a batch experiment also depends on the material-to-solution ratio. This was observed for shell sand in a study showing that a 1:1 ratio with 25 mg P L⁻¹ reached equilibrium after 24 h while a 1:15 ratio did not (Søvik and Kløve, 2005). In another study, the P removal efficiency of Filtralite P increased considerably after 48 h compared with that after only 24 h (Ádám et al., 2007b). The effect of the material-to-solution ratio was also investigated in that study and the conclusion reached was that smaller material-to-solution ratios need longer time to reach equilibrium and therefore batch experiments using small material-to-solution ratios may need a contact time of more than 24 h to give reliable measurements of maximum P sorption.

The agitation has to be fast enough to ensure mixing without causing breaking-up of the material. In general, speeds between 100 and 200 rpm seem to be appropriate according to the selected studies (Table 2), but some materials can break or disintegrate at speeds above 100 rpm (personal observation), thus altering the properties of the material (initial particle size and porosity) and most likely overestimating the PSC. Phosphorus sorption on BFS has been found to increase with increasing agitation by enhancing the diffusion of phosphate ions into the pores of the material (Oguz, 2004).

Temperature can affect sorption and it is important to specify the temperature at which the experiment is performed (Table 2). Room temperatures ranging from 20 to 25°C commonly give similar results. However, P sorption increases at higher temperatures, as it has been shown for fly ash and furnace slag by Agyei et al. (2002). The effect of temperature on P sorption has also been studied by Ugurlu and Salman (1998). They contributed to understanding the effects of different batch parameters on the sorption capacity of fly ash.

Adsorption Isotherms

The relationship between the equilibrium concentration of a solute in solution (C_{eq}) and the amount of solute sorbed to the material (S) at constant temperature is referred to as the adsorption isotherm (Stumm and Morgan, 1996). Adsorption

isotherms can be described mathematically with the general formula (Barrow, 1978; Sposito, 1989; McKay, 1996):

$$S = (K \times C_{eq}) / (A + B \times C_{eq}^D) \quad [3]$$

where K , A , B , and D are isotherm constants.

A number of isotherms have been developed over the years based on different assumptions that simplify Eq. [3]. The Langmuir isotherm was proposed for homogeneous adsorption and it assumes a uniform adsorbent surface with energetically identical sorption sites (Langmuir, 1919; Olsen and Watanabe, 1957). The Langmuir equation is obtained by setting $A = B = 1$ in Eq. [3] and has the form:

$$S = S_{max} \times [(K_L \times C_{eq}) / (1 + K_L \times C_{eq})] \quad [4]$$

where K_L is the Langmuir constant and is related to the energy of adsorption and S_{max} is the maximum adsorption capacity. By linearizing Eq. [4], the Langmuir parameters K_L and S_{max} can be obtained from the slope and intercept of plotting the inverse sorption and equilibrium concentrations.

The Freundlich equation describes equilibrium on heterogeneous surfaces and is expressed as:

$$S = K_F \times C_{eq}^{b_F} \quad [5]$$

where K_F and b_F are the Freundlich constants, K_F expresses the adsorption capacity (the larger the value, the higher the capacity) and b_F is the heterogeneity factor. The Freundlich isotherm is widely recommended because of its accuracy and it has been often found to describe P sorption better than Langmuir (Barrow, 1978; Sakadevan and Bavor, 1998; Zhou and Li, 2001). However, it does not allow a maximum sorption capacity to be predicted. Although less used, there are many other isotherm equations such as the Temkin equation (Barrow, 1978), the Redlich-Peterson isotherm (McKay, 1996), the Frumkin equation (Stumm and Morgan, 1996), and different modifications and kinetic versions of the Langmuir and Freundlich isotherms (McGechan and Lewis, 2002).

In the selected papers, Langmuir isotherms appear to be the most popular to explain P sorption, but Freundlich isotherms are also used (Table 2). The Langmuir maxima give an estimate of the PSC (Mann and Bavor, 1993; Cheung et al., 1994; Sakadevan and Bavor, 1998; Kostura et al., 2005; Li et al., 2006; Xu et al., 2006; Boujelben et al., 2008). In some cases, the experimental data are not fitted to any model and the sorption capacity at the highest initial P concentration is used (Zhu et al., 1997; Johansson, 1999; Ádám et al., 2007a).

Limitations of Adsorption Equations

The use of isotherm equations is sometimes nonviable because the data simply do not fit to the model or the model cannot explain the results. For example, in a batch test with fly ash and furnace slag the Langmuir and Freundlich equations did not fit to the experimental data while the Frumkin equation did (Agyei et al., 2002). Langmuir and Freundlich isotherms appear to be linear at low concentrations (Stumm and Morgan, 1996) and behave quite differently at higher concentrations. Furthermore, adsorption isotherms are used to describe sorption processes that may involve re-

actions other than adsorption. The effect of precipitation reactions on the shape and applicability of commonly used isotherms such as Langmuir and Freundlich has been studied by many authors (Sakadevan and Bavor, 1998; Arias et al., 2001; Zhou and Li, 2001; Søvik and Kløve, 2005). A recent study showed that Langmuir isotherms accurately described P sorption by sands if no precipitation reactions were taking place (Del Bubba et al., 2003).

The application of the Langmuir equation is restricted by all the assumptions involved in it and the use of the sorption maxima (S_{max}) must be viewed with caution (Barrow, 1978; Stumm and Morgan, 1996; Drizo et al., 2002). The Langmuir equation implies a 1:1 stoichiometry and proportionality between the activities of the surface species and their concentrations. The conditions for validity of a Langmuir-type adsorption equilibrium are: (i) equilibrium up to the formation of a monolayer; and (ii) equal activity of all surface sites (Stumm and Morgan, 1996). Although these conditions may not always be achieved, some authors have observed a good correlation of the experimental data to the Langmuir model (Kostura et al., 2005; Xu et al., 2006; Boujelben et al., 2008). The fit of experimental data to a Langmuir (or other) adsorption isotherm does not constitute evidence that adsorption satisfies the criteria. Frequently, adsorption to a surface is followed by additional interactions at the surface or within the matrix of the material.

Different studies have pointed out the limitations of using Langmuir isotherms to describe P sorption by filter materials. Drizo et al. (2002) observed significant differences in the Langmuir constant and adsorption maxima for EAF slag when using different initial P concentrations. The study reported a 13-fold increase in the adsorption maxima and 830-fold decrease in the bonding capacity within the range of concentrations applied. In another study, the Langmuir maxima did not correspond to the sorption capacity estimated in column studies for sands (Arias et al., 2001). The Langmuir equation should only be used to calculate the maximum adsorption capacity when the range of P concentrations added to a soil accurately reflects the range of P in the system to be modeled (Mann, 1996). While the Langmuir equation can underestimate the sorption capacity of soils, it usually overestimates the sorption capacity of reactive materials. This was observed in a comparison of batch experiments and large-scale filter systems using shell sand (Søvik and Kløve, 2005).

Alternatives to Adsorption Isotherms

The graphical representation of S from different initial P concentrations against C_{eq} (or C_0) can give an idea of the sorption behavior and in many cases shows a saturation curve that reaches a point at which increasing concentrations do not cause an increase in the amount sorbed (S). This saturation point can be used to directly estimate the maximum sorption capacity of the material. In other cases, however, the graphical plot gives a logarithmic curve or even a straight line that indicates no saturation and makes prediction of the sorption capacity graphically impossible. This can be observed in some of the selected studies (Drizo et al., 1999; Johansson, 1999; Xu et al., 2006; Ádám et al., 2007a). In such cases the experimental set-up may be rearranged to fully saturate the material. Maximum sorption normally occurs at the highest initial P concentration and

thus some authors take that value as the sorption capacity of the material (Table 2). However, P removal efficiency depends strongly on the concentration applied, which has been found to fluctuate considerably among studies. For example, while Zhu et al. (1997) estimated the sorption capacity at 320 mg P L⁻¹, Johansson (1999) estimated it at 25 mg P L⁻¹ (Table 2).

Thus, as an alternative to adsorption isotherms, the sorption capacity can be simply estimated as the amount of P sorbed to a material from a sole initial P concentration. This is also known as the single point P sorption index at a given initial P concentration. Attempts to standardize a procedure to estimate the PSC of soils have yielded several tests with defined soil-to-solution ratio, P concentration, contact time, and other parameters (Mann, 1996; Patterson, 2001). Some authors have proposed a classification system according to the P sorption index. For instance, soil analysis guidelines proposed five classes of soils based on the percentage of P retention from a single addition of 1000 mg P L⁻¹ (Patterson, 2001). A previous study related the mineralogy of soils to the PSC, also defining five classes and giving those soils with a PSC over 1 g P kg⁻¹ a "very high" rating (Mann, 1996).

However, many studies have shown a clear discrepancy between the single point sorption index and the Langmuir maxima (Arias et al., 2001; Patterson, 2001). On the other hand, some authors have shown a good correlation. This is the case for the one-point isotherm used to evaluate the sorption capacity of calcareous soils and limestone (Zhou and Li, 2001). Another method developed for aluminum-based water treatment residuals (WTR) showed a good correlation between a modified single point ammonium oxalate-extractable aluminum (Al_{ox}) and the Langmuir maxima (Dayton and Basta, 2005).

Interpretation of Batch Experiment Data

The present study has shown how vulnerable the results can be to batch experimental parameters, and, therefore, how difficult to compare the data indeed is. To minimize the effect of such parameters and obtain more accurate results, the experimental procedures should reduce discrepancies. Graetz and Nair (2000) presented a set of guidelines for P sorption isotherm determination in soils based on the investigation of Nair et al. (1984). Based on these guidelines, and the analysis of batch experiment procedures in the selected studies, recommendations on main batch experiment parameters shall include:

- Particle/aggregate diameter of the same order of magnitude (the materials could be classified according to their particle size and hydrological properties).
- Similar material-to-solution ratio, as close as possible to that use in soil science (1:20), and, preferably, similar amount of the material. Graetz and Nair (2000) proposed an amount of 0.5 to 1 g for soils. For filter materials a range between 1 and 3 g may be appropriate. In case of very fine materials, 0.5 or even 0.25 g could be used.
- P concentrations range close to that of wastewater or slightly higher (max. 100–200 mg P L⁻¹).
- Contact times of at least 24 h (preferably longer). Preliminary kinetic studies are recommended.

- Agitation of about 100 rpm, so that the solution moves but not the material.
- Temperature between 21 and 25°C (room temperature)

Classification of Filter Materials

There is a large variation in filter materials and they can be classified according to different parameters such as origin, mineralogy/structure/chemical composition, particle size distribution, etc. This may help in assessing the applicability of a material to a specific filter system. Another important factor from the practical point of view is the efficiency of the material and therefore a classification according to the PSC may be of great interest to provide criteria for material screening. Based on the results from the selected and other studies, filter materials were classified into five different groups according to their PSC (Table 4). At the same time, the materials were divided into two categories according to their particle size distribution. These two categories separate fine and coarse materials under and above a particle diameter of 1 mm.

Gravels and some soils have the lowest PSC. Fly ash and slags can have from moderate to very high capacities depending on their composition and particle size. The commercial products Filtra P and Polonite have a high to very high PSC based on data from several studies using different methodology to estimate the sorption capacity (Brogowski and Renman, 2004; Cucarella et al., 2007; Gustafsson et al., 2008). There are many other materials not included in the table. Some of the materials with reported high to very high capacities are volcanic ash, with about 10 g P kg⁻¹ (Ping and Michaelson, 1986); half-burned dolomite grain, with 20 g P kg⁻¹ (Roques et al., 1991); ochre, with up to 30 g P kg⁻¹ (Heal et al., 2005); and Tobermorite, which has been recently shown to yield a P-rich product (with up to 13% P) that could be used as a substitute of phosphate rock (Berg et al., 2005, 2006).

Estimation of the Lifetime of Filter Materials

Both the PSC and particle size are parameters of great importance when selecting a suitable material for a filter system (Sakadevan and Bavor, 1998; Drizo et al., 1999). Based on the PSC estimated in batch experiments, some authors have ventured to predict the lifetime of filter materials used in wastewater treatment systems such as constructed wetlands (Table 3). Sakadevan and Bavor (1998) concluded that a conventional constructed wetland (CW) system using BFS could last for 58 yr. However, as discussed earlier, their batch experiment conditions were far too different to field conditions. Drizo et al. (1999, 2002) suggested a lifetime of a CW system of 7 yr using shale and 13 to 37 yr using EAF slag based on the results of column studies. Another study suggested a potential lifetime of a CW system using furnace slag of 22 yr based on the Langmuir P maxima (Xu et al., 2006). At the same time, this and other studies recommended full-scale supplementary research to accurately forecast the performance of filter materials under field conditions.

The performance and lifetime of on-site wastewater treatment facilities using filter materials depend mainly on the material used, and therefore on its structure and particle size, chemical composition, and pH (Mann, 1996; Sakadevan and Bavor, 1998). However, there are other important factors such as hydrological condi-

Table 4. Classification of the filter materials investigated in the selected studies according to the P sorption capacity and particle size.

P sorption	P sorption capacity g P kg ⁻¹	Filter materials	
		Fine (<1 mm)	Coarse (>1 mm)
Very low	<0.1	Soils	Gravels
Low	0.1–0.5	Sand, Soils	LECA‡, Limestone, Opoka
Moderate	0.5–1	Bentonite, Calcareous soils, Fly ash†, Spodosol	Bauxite, BFS†, Zeolite†
High	1–10	BFS†‡, Fly ash†, Fe-coated sand and brick	BFS†, EAF‡, Filtra PS, Filtralite P, Polonite§, Shell sand, UTELITE
Very high	>10	BFS†, Fly ash†, Polonite§, Red mud	n.d.‡

† Depending on chemical composition.

‡ BFS, blast furnace slag; EAF, electric arc steel furnace slag; LECA, light expanded clay aggregates; n.d., no data.

§ Based on other studies (Brogowski and Renman, 2004; Cucarella et al., 2007; Gustafsson et al., 2008).

tions, organic matter, ionic strength and salinity, redox potential, competing ions, temperature, and microbiological transformations (Mann, 1996) which may impact some of the dominant processes and are not taken into account in batch experiments. The more components in the system, the more complicated it becomes to evaluate the performance of the filter material and predict its lifetime. For instance, a constructed wetland with vegetation open to climatic conditions will certainly have the influence of too many parameters to simplify P sorption to a batch test model. On the other hand, more simple systems such as filter wells, which consist of a compact column filled with a reactive filter material, may behave more similarly to laboratory estimates. For such systems, the combination of sorption models derived from batch experiments with other models (kinetic, hydrological, etc.) may be a reliable tool to predict the efficiency and lifetime of the system.

Conclusions

The filter materials investigated in a number of selected studies vary considerably in origin, particle size, chemical composition, and pH. There is a visible relationship between the estimated PSC of the materials investigated and their particle size and pH. There is a stronger correlation between P sorption and some of the batch parameters, that is, initial P concentration and material-to-solution ratio, suggesting that batch parameters have great repercussions for the results.

The present comparison shows that batch experimental parameters are arbitrarily established and vary significantly among studies. Large discrepancies exist in the form and amount of the material used, material-to-solution ratio, P concentration range and extent of agitation. These differences render it impossible to compare the capacity of the materials and question the validity of the results.

Batch parameters should be appropriately established to minimize their influence on P sorption. Procedures should be standardized for materials within the same particle size range, using similar material-to-solution ratios and applied P concentrations, contact times that allow equilibrium to be reached (based on previous kinetic studies), proper agitation to ensure mixing but avoid aggregate breakage, and constant room temperature.

Filter materials can be classified according to their PSC into five groups. The classification further subdivides the materials into two categories according to their particle size. The classification system is a helpful tool for material screening and may be used to categorize new filter materials.

The PSC determined in batch experiments may be useful to predict the lifetime of filter materials if the experimental set-up accurately represents the field conditions and the data are treated appropriately. Further research is needed to develop models including batch experimental data, thus providing a more reliable tool that would allow to more accurately estimating the lifetime of filter materials used for on-site wastewater treatment.

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